Assessing marginal abatement cost for greenhouse gas emissions from livestock production in China and Europe - accounting for uncertainties

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Declaration

I hereby confirm that this submitted doctoral dissertation is an original research work conducted by me and that none of this work has been submitted for any other degree or professional qualification. All work described in this dissertation is my own except where stated otherwise.

This research study was carried out in partial collaboration with several scientists. I hereby confirm that appropriate credit has been given to the contribution of other authors for chapters that were or will be submitted in peer-reviewed journals. In these papers, the candidate appears as the first author and has made significant contribution to these papers. Contribution of co-authors is stated below:

Chapter 4 is partly based on research results from a study conducted by Dr. Wen Wang (Chinese Academy of Agricultural Sciences) and myself as joint first authors (see Wang et al., 2014).

Chapter 5 is based on a joint paper intended for submission. Dr. Jan Peter Lesschen (Alterra Wageningen UR) introduced me to the MITERRA-Europe model and helped me with the model simulation.

Chapter 6 is based on a joint paper intended for submission. Dr. Michel de Haan (Wageningen UR Livestock Research) supplied estimates of cost-effectiveness for some mitigation options and Dr. Abdulai Fofana (Scotland's Rural College) introduced me to Vose ModelRisk 4.3.

For Chapter 7, Dr. Adam Butler (Biomathematics and Statistics Scotland) assisted me in using Kernel Density estimation in the statistical program R.

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Abbreviations

AD-E	On-farm anaerobic digestion with electricity generation
AD-H	On-farm anaerobic digestion with heat production
BAU	Business as Usual
C	Carbon
CAPRI	Common Agricultural Policy Rationalised Impact
CAPSiM	China's Agricultural Policy Simulation Model
Cdf	Cumulative density function
CE	Cost-efficiency
CEA	Cost-efficiency analysis
CGE	Computable general equilibrium models
CH_4	Methane
CI	Confidence interval
CO_2	Carbon dioxide
CO_2e	Carbon dioxide equivalent
COP	Conference of Parties
CT	Condensed tannins
DCD	Dicyandiamide
DG-AGRI	Directorate-General for Agriculture and Rural Development
DM	Dry matter
DMI	Dry matter intake
EF	Emission factor
ENG	Engineering
EPPA	Emission Prediction and Policy Analysis
ETS	European trading scheme
EU	European Union
EU-15	European Union 15
FADN	Farm Accountancy Data Network
FAO	Food and Agriculture Organisation
FAPRI	Food and Agricultural Policy Research Institute
FPCM	Fat and protein corrected milk
FSS	Farm structure survey
GAINS	Greenhouse Gas and Air Pollution Interactions and Synergies
GDP	Gross domestic product
GHG	Greenhouse Gas
Gt	Gigaton
GWP	Global warming potential
Ha	Hectare
IFCN	International Farm Comparison Network
IPCC	International Panel on Climate Change
KDE	Kernel density estimation
Kt	Kiloton
L	Litre
LCA	Life cycle analysis

LGI	Light grazing intensity
LHS	Latin Hypercube sampling
LUC	Land use change
M	Million
MAC	Marginal abatement cost
MACC	Marginal abatement cost curve
MC	Monte Carlo
MDC	Marginal damage cost
MGI	Medium grazing intensity
MSB	Marginal social benefit
MSC	Marginal social cost
Mt	Megaton
N	Nitrogen
N_2	Nitrogen gas
N_2O	Nitrous oxide
NH_3	Ammonia
NH_4^+	Ammonium
NITR	Soil nitrification inhibitors
NO_2^-	Nitrite
NO_3^-	Nitrate
NOTILL	No tillage
NO_x	Nitrogen oxides
NPP	Net primary production
NPV	Net present value
NUTS	Nomenclature of Territorial Units for Statistics
NVZ	Nitrate vulnerable zone
OECD	Organisation for Economic Cooperation and Development
Pdf	Probability distribution function
REDFERT	Reduced amount of mineral fertiliser application
PEM	Partial equilibrium models
PRNG	Pseudorandom number generator
SA	Scenario analysis
SAPM	Survey on agricultural production methods
SOC	Soil organic carbon
SPLITFERT	Multiple application of fertiliser
SSM	Supply side models
SU	Sheep unit
T	Ton
TIMEFERT	Improved timing of fertilisation
UAA	Utilised agricultural area
UNFCCC	United Nations Framework Convention on Climate Change
USA	United States of America

Abstract

Climate change is probably the most challenging threat to mankind. International agreements have acknowledged the fact that anthropogenic GHG emissions must be reduced significantly to adhere to a maximum global warming of 2°C. The livestock sector plays a key role in achieving this target as it is a significant source of GHG emissions. While the livestock sector offers significant GHG reduction potential, it is currently neglected in international and national mitigation efforts. Therefore, scientific research must guide mitigation policy decisions with evidence of cost-efficient abatement potential that can be achieved through various mitigation technologies.

Marginal Abatement Cost Curves (MACC) are an analytical tool for informing policy makers about the cost-effectiveness (CE) of mitigation. MACCs provide a relatively clear representation of a complicated issue based on their graphical design that prioritises various mitigation options in terms of their CE of abatement and enables assessment of total GHG reduction under a budget constraint. However, developing a MACC involves considerable data collection, depends on various interdisciplinary information sources and the methodology is subject to several limitations. These factors can result in uncertainties in marginal abatement cost (MAC) results, the assessment of which is often neglected in MACC literature.

This research shows the main GHG emission sources in livestock production and possible mitigation options to reduce GHG emissions from these sources. After elaborating the MACC methodology, advantages, disadvantages and limitation of the engineering MACC are shown. This allows understanding the relevance of assessing and reporting uncertainty of MACCs. Two engineering MACCs are developed that show the CE abatement potentials available in the Chinese livestock sector and European Union 15 (EU-15) dairy sector in 2020, with emphasis on dietary mitigation options. The requirement of assessing CE of abatement for individual mitigation options is highlighted by separate derivation of technical and economic abatement potential for the EU-15 dairy sector. For the Chinese MACC, a scenario analysis (SA) and for the European MACC, a Monte Carlo (MC) simulation are utilised to show the relevance of assessing uncertainty in MACCs. To provide further evidence, the overall range of CE estimates for eight mitigation options found in relevant MACC literature is presented. This allows the generation of probability distribution functions of CE for each mitigation option with kernel density estimation (KDE).

The results from this study show the significance of livestock and dairy production related GHG emissions in China and Europe, respectively. In China, baseline GHG emissions of livestock production are projected to increase significantly, while these of the EU-15 dairy production are predicted to decrease by 2020. It was found that enteric fermentation is the largest GHG emission source from dairy production and should be focus of mitigation policies. Both case studies showed mitigation options that offer abatement potential at high CE. Priorities should be given to biomass gasification, breeding techniques and feed supplements as tea saponins and probiotics for the Chinese livestock sector, and to animal selection, reduced tillage and dietary probiotics for the EU-15 dairy sector. The scenario analysis reveals that mid-term projections for the Chinese livestock sector are varying strongly, and utilising key variables from different projections has a significant impact on MAC results which changes the ranking of the mitigation options. The MC simulation shows the contribution of some model inputs to the uncertainty of abatement at negative cost and a high model output uncertainty regarding measure's CE for most mitigation options. However, the ranking of the mitigation options remains stable. The range of MAC estimates for 8 mitigation options in the agricultural sector is high and variables like 'study quality' or 'study location' do not change this. The KDE was further used to rank the mitigations options based on their probability of being reported as cost-negative and shows that measures affecting soil N_2O and carbon sequestration are reported to be more cost-efficient as compared to measures focusing on manure management. Based on these finding, the impact of study designs on MAC estimates and lack of communication uncertainty in MACC literature are discussed.

Uncertainties that are underpinning MACC results can have significant impacts on CE and abatement potentials. To increase utilisation of MACCs by knowledge users, MACC research must prioritise assessment, quantification and report of uncertainties, compare results within the scientific literature and publish data and assumption of the MACC transparently.

Lay summary

Climate change is probably the most challenging threat to mankind. As climate change is triggered by increasing levels of GHG emissions, policy makers should implement policies that are targeted at reductions in GHG emissions. The livestock sector is a significant contributor to total anthropogenic GHG emissions. Despite offering large GHG reduction potentials, its role in reducing global anthropogenic GHG emissions has not been considered at national and international levels for reducing GHG emissions. Therefore, science must extend to the assessment of cost-efficient reductions available in the livestock sector and highlight this information to policy.

Marginal Abatement Cost Curves (MACC) are an analytical tool that allows policy makers to understand the GHG reduction potential and cost-effectiveness of different mitigation options i.e. technologies that are implemented to reduce GHG emissions. Developing a MACC is a complicated procedure and the methodology has several limitations. Biophysical and economic uncertainties are a significant issue in relation to mitigation measures and most MACC studies have so far not adequately assessed and reported these uncertainties. This study focuses on the European Union (15) dairy and Chinese livestock sectors as both of those systems are major GHG emission sources. Through development of a MACC for each region, the available cost-efficient GHG reduction potential for each region has been shown. Since GHGs emitted directly by the animal are the largest source of GHG emissions in ruminant livestock systems, this study emphasised on mitigation measures that focus on this source i.e. feed additives. To highlight the importance of assessing the cost-efficiency of GHG reduction, in the context of the European case study, firstly the technical abatement potential i.e. total GHG reduction was estimated. Thereafter the economic GHG reduction potential i.e. GHG reduction available at a certain cost threshold was assessed. Uncertainties were assessed for the Chinese case study via scenario analysis and for the European MACC via Monte Carlo simulation. Additionally, the MACC literature was reviewed and the overall range of reported cost-efficiencies for eight different mitigation options was shown.

The results of this study show the significance of livestock and dairy production related GHG emissions in China and Europe, respectively. In China, GHG emissions from livestock were expected to increase significantly until 2020. Some mitigation options offered GHG reduction potential that is available at low costs, and should therefore be prioritised by the policy makers. The uncertainty assessments revealed that MACC results can be uncertain. The scenario analysis showed that mid-terms projections for China varied strongly and some

key assumptions had a strong impact on the MACC results. The Monte Carlo simulation showed high uncertainty for most of the mitigation options. Based on the MACC literature review, it was found that the range of reported cost-efficiencies is large and this may be due to uncertainties in MACC assessment.

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Chapter 1 – Introduction

1.1 Climate change and international mitigation efforts

Climate change is probably the biggest threat to mankind with irreversible impacts on social, economic and ecological systems globally (Stern, 2007). Rising global temperatures are leading to oceanic acidification, changes in global water cycles, sea level rise, and an increased frequency of heat waves, floods, droughts, storms, infectious diseases and heavy precipitation events (IPCC, 2014). Policy makers are aware of these dramatic consequences and acknowledge the anthropogenic causes of climate change. Over the last four decades, increasing efforts for mitigation and adaptation to climate change have been undertaken at both national and international levels. A key driver for this development is the generally accepted agreement to limit global temperature increase to a maximum of 2°C relative to pre-industrial levels as proposed by the United Nations Framework Convention on Climate Change (UNFCCC; Rosen and Guenther, 2015). In order to achieve this target, an immediate and significant cut of anthropogenic greenhouse gas (GHG) emissions is required for reducing climate change impacts and minimising mitigation costs (Meinshausen et al., 2009; van Vliet et al., 2012). Several meetings and follow-up agreements at international level have addressed this issue, culminating at the most recent Conference of Parties (COP-21) in 2015 in Paris. During the seventh COP in 2001, detailed rules of the Kyoto Protocol were set, including an average GHG reduction target of 5% below 1990 levels for ANNEX I countries during the first commitment period between 2008 and 2012 (Halkos, 2014). The Kyoto protocol came into force in 2005 with 37 participating industrialised countries and the European Community. In 2012 during the 18th COP, the Kyoto Protocol was extended with a second commitment period from 2013 to 2020 targeting an average GHG emissions reduction of 18% below 1990 levels (Halkos, 2014). There is a general agreement on seeking cost-efficient GHG reductions which also highlights the importance of developing national and international carbon trading schemes. Europe and China are significant contributors to total anthropogenic GHG emissions and should therefore be prioritised in the global climate change mitigation agenda. Their ambitious GHG reduction targets make it particularly important to seek cost-efficient GHG reduction measures throughout the economy. In line with the international climate change mitigation efforts, the European Union (EU) and China declared their own ambitious GHG reduction targets. For the EU, these targets were partly driven by a GHG reduction target of 8% relative to 1990 levels during the first commitment period of the Kyoto Protocol. In 2007, the EU endorsed the European Climate-Energy package which set the 20-20-20 targets, thereby aiming for a GHG emissions reduction of 20% below 1990 levels, a contribution from renewable energy sources to total energy production of 20% and a reduction of primary energy usage by 20% in 2020. In 2009, the EU had decided on a 10% reduction in GHG emissions from non-European Emission Trading Scheme sources until 2020 as compared to 2005 levels (De Cara and Jayet, 2011). In a more recent agreement, the EU endorsed a GHG reduction target of 40% below 1990 levels by 2030. China was not listed as ANNEX I country by the UNFCCC during the first commitment period and consequently did not ratify binding GHG reduction targets. In 2009 during the COP-15 (known as the Copenhagen Accord), China presented its first voluntary climate change mitigation commitments. Key elements of the announcement included a carbon intensity reduction by 40-45% per produced unit of Gross Domestic Product (GDP) relative to 2005 levels, 15% energy supply from non-fossil fuels and a national forest cover of 40 million hectares (M ha) by 2020. During the 17th COP in 2011, China announced its intention to participate in an internationally binding post 2020 GHG reduction agreement, subject to the complement of certain conditions. In line with these announcements, climate change mitigation targets were introduced in the 12th Five-Year Plan that covered the period from 2011 to 2015. China targeted a carbon intensity reduction of 17% per produced unit of GDP by 2015 relative to 2005 levels and to produce 9.5% of the energy mix from renewable energy systems.

In spite of these international and national efforts, the International Panel on Climate Change (IPCC) reported in 2014 that annual anthropogenic GHG emissions increased between 2000 and 2010 as compared to the period from 1970 to 2000, with an average annual growth of 2.2% (IPCC, 2014). Main reasons for this are global population and economic growth (IPCC, 2014). However, for the EU including 27 member countries, GHG emissions were reduced by 0.72 Gigaton carbon dioxide equivalent (Gt CO_2e; for EU-15 by 0.34 Gt CO_2e) between 1990 and 2010. China did not show such promising development as total GHG emissions increased by 6.9 Gt CO_2e during the same period (Edenhofer et al., 2014), thereby surpassing the USA as the world's largest GHG emitter.

1.2 Livestock activities and climate change

Agriculture is the most essential human activity providing employment and livelihoods for a large share of human population, particularly in developing countries. However, agricultural production has been both a driving factor for triggering as well as being affected by climate change. The sub-sector 'animal agriculture' emits up to 18% of total GHG emissions if the whole production lifecycle including land use change (LUC) for pastures and feed production, production of fertiliser, organic and inorganic fertiliser application, land degradation, fuel combustion for production, transport and processing of animal products is considered (Gill et al., 2010). Despite livestock production being the main emitter of total methane (CH_4) and nitrous oxide (N_2O), both of which are key GHG (Gerber et al., 2013a), this sector is responsible for further adverse production externalities such as eutrophication of water bodies, biodiversity loss, air pollution, increased water consumption and LUC associated with deforestation (Herrero and Thornton, 2013; Steinfeld et al., 2006). However, livestock sector's vulnerability to climate change can have serious implications for food security and livelihoods of millions of people. Several factors contribute to this vulnerability. First, livestock production is a major consumer of natural resources e.g. feed from agricultural production and water for drinking, growing crops, servicing and product processing (Thornton et al., 2009). Meanwhile climate change has negative impacts on average crop yield and fresh water availability in some regions (IPCC, 2014). Second, heat stress for the animals has negative effects on animal yield and health (Nardone et al., 2010). Third, higher temperature and precipitation trigger larger populations of pathogens and vectors of diseases (Thornton et al., 2009). Finally, the frequency and intensity of extreme weather events can have negative impacts on livestock production. It is likely that livestock systems in developing countries particularly small scale and pasture based systems are more strongly affected by climate change (Nardone et al., 2010), as these systems generally show low adaptation capacities as compared to intensive livestock production in housing facilities that are more often prevalent in the developed world.

Between 1963 and 2003, global demand for livestock increased significantly (Kearney, 2010) and thereby GHG emissions and other production externalities correspondingly. With an expanding global human population that may reach 9 billion people by 2050 and with 70% of these inhabiting urban regions (Marchal et al., 2011), it is expected that demand for livestock products and hence livestock production externalities will further increase. A major challenge for this sector is to reduce its GHG emissions output while simultaneously increasing its production levels to reduce the carbon footprint per produced unit of animal protein, meet the increasing demand and compensate for potential production losses through climate change. Although efficiency of livestock production increased in many livestock systems globally, total GHG emissions also increased (Edenhofer et al., 2014) and are expected to further increase in future. The global livestock sector shows an impressive technical GHG reduction potential that can be 30% of its total emission output, if livestock producers operating within the same system region and agro-ecological zone adopt the management techniques of the 10% most production efficient farms (Gerber et al., 2013a). It should be noted that GHG reduction potentials vary strongly across different regions depending on bio-physical, climatic, social and technological settings with livestock systems in developing economies being less efficient as compared to developed economies. Therefore, GHG reduction through efficiency gains might be larger there.

Despite an immediate requirement for GHG reduction in every economic sector to adhere to a maximum global temperature increase of 2°C, the livestock sector is often neglected in national and international climate change mitigation efforts. Reasons for this include failure to establish an internationally binding post-Kyoto agreement, global economic downturn that reduces the willingness of governments to impose GHG regulations, and a lack of information on CE of mitigation options for the livestock sector. For instance, out of the 40 industrialised countries listed at the UNFCCC, only Bulgaria and France enforced legislation to target GHG reduction in the livestock sector in quantitative terms (Bailey et al., 2014). GHG reduction in this sector through policy intervention is mainly achieved indirectly through non-climate policies.

Although scientific literature estimated the technical GHG reduction potential for the livestock sector at both global level and for individual countries, scientific knowledge is lacking with regards to the economic abatement potential offered by the livestock sector. Assessing the economic abatement potential is particularly important for mitigation policy design as this allows conclusions as to whether the livestock sector offers a cost-efficient abatement potential or not. However, such assessment is particularly difficult for this sector in which operations of many small scale producers are being characterised by high heterogeneity in terms of economic and bio-physical settings. The identification of appropriate mitigation technologies applicable throughout the sector must therefore consider these characteristics. Lack in CE assessments make mitigation policy design in this sector a gamble, and could lead to enforcement of mitigation policies that induce a high cost burden for the livestock producer. This subsequently could reduce production levels and/or competitiveness in a global food market. This would contradict the target of self-sufficiency in food production for many countries and diminish the maintenance of livelihoods of many people. Therefore, it is important to develop research on climate change mitigation economics in the livestock sector to guide mitigation policies for reducing GHG emissions in this sector cost-efficiently. MACCs have been found to be an appropriate tool for identifying CE of mitigation options. These curves can inform policy makers about available technologies that can lead to desirable GHG reduction under budget constraints while reporting GHG abatement potentials and associated costs of these technologies (Halkos, 2014). MACCs can be used at different scales as policy makers require information on cost of mitigation for larger regions. Although this implies a certain level of generalisation in terms of the applicability of measures. While it is useful to understand the aggregated effectiveness of mitigation efforts, it is important to develop potentials based on more granular analysis of what works on speciefic farms. Hence bottom up- analysis is required. However, MACC studies have been critised for neglecting the involved input and output uncertainties. This is particularly important for this tool as it depends on a large set of interdisciplinary data and includes economic projections that may be subject to uncertainty. Furthermore, the cost-efficiency estimates justify action or inaction for climate change mitigation and can have major implications ~~to~~ on the economy and the planet. Lack of uncertainty assessment can be a key reason for failure of MACC prediction and could result in undesirable outcomes after enforcing mitigation policies. Uncertainty assessment of the MACC exercise is therefore crucial for increasing awareness on potential errors, inform decision makers accurately and increase robustness of decisions based on the MACC outcome (Lempert and Schlesinger, 2000).

1.3 Aim of this research

This dissertation focuses on assessing the abatement potential and CE of GHG reduction for various mitigation options that are applicable to livestock production. Emphasis is placed on dietary technologies as GHG emissions from enteric fermentation are the main GHG emission source from livestock production. For this purpose, two individual MACCs have been developed that target two different regions i.e. China and EU-15 which are exemplary of globally significant livestock production systems. Focus on China is particularly important due to its role as a growing economic superpower with rising GHG emission output which increases its responsibility towards global climate change mitigation. The livestock sector remains an untapped opportunity for GHG emission reduction in China, particularly in the light of the ambitious GHG intensity reduction targets. The second case study focuses on the EU-15 dairy sector. Europe traditionally plays an important role in global dairy production, representing 25% of global milk production (Lesschen et al., 2011); hence the dairy sector is responsible for a significant share of livestock GHG emissions in Europe. While global demand for dairy products increases, Europe as a main exporter of dairy products has to increase production levels while reducing GHG emission to adhere to ambitious GHG reduction targets. This dissertation also focuses on the uncertainties that are involved in MACCs and presents their significant impact on MAC results. SA and MC simulation are utilised as exemplary uncertainty assessment tools for understanding this impact and to increase awareness of the requirement of uncertainty assessment to enable MACCs to correctly guide policy decisions. To further emphasise this issue, this study focuses on the overall range of CE estimates reported for the most common mitigation options as found in relevant MACC literature and thereby elaborating on how study design impacts MAC estimates.

1.4 Structure of the dissertation

This dissertation is structured as follows: Chapter 2 depicts the sources of GHG emissions from livestock production, quantifies GHG emission output from Chinese and the European agricultural sectors and introduces a range of mitigation technologies applicable to livestock production. Chapter 3 highlights the need for assessing CE of GHG abatement from different mitigation options and thereby elaborates the differences between technical and economic abatement potential. This serves as an introduction to a detailed elaboration of the MACC tool including various MACC approaches and shortcomings of these MACC approaches with emphasis on uncertainties during the MACC exercise and how to handle these. Chapter 4 describes the first case study i.e. the development of a MACC for the Chinese livestock sector including a SA. Chapter 5 elaborates the second case study while estimating the technical abatement potential for the EU-15 dairy sector under consideration of some GHG emission sources outside the farm gate. Based on these findings, in chapter 6**,** the development of a MACC for the EU-15 dairy sector is described, along with the assessment of input and output uncertainties involved in this exercise via a MC simulation. Chapter 7 reviews the MACC approach based on a meta-analysis showing the range of MAC estimates for eight mitigation technologies, and elaborates characteristics of different MACC studies that may lead to varying CE estimates for mitigation options.

Based on the findings of the previous chapters, this dissertation concludes on: i) the economic abatement potential that is offered by the livestock sector in China and EU-15, ii) the importance of feed additives in climate change mitigation in livestock production, iii) uncertainties of MACC results and implication on policy decision support and iv) guides future research to strengthen the MACC approach.

2 Chapter 2 - Literature review

2.1 Sources of GHG emissions in livestock production

To assess GHG reduction potentials within the livestock sector, it is important to understand GHG emissions sources and quantify GHG emissions from livestock production. Livestock production activities emit GHG emissions that can occur within or outside the farm gate. Those GHGs within the farm gate include CH_4, N_2O and carbon dioxide (CO_2) and are known as major GHGs considering their contribution to climate change (Steinfeld et al., 2006). Primary sources of these GHG emissions are enteric fermentation, agricultural soil (including grasslands and cropland) and manure/slurry management. While enteric fermentation and manure management can be solely attributed to livestock activities, GHG emissions from agricultural soil are only partly caused by feed production for livestock. In the following sections, these potential GHG sources are discussed in detail. Emissions outside the farm gate can be attributed to a wide range of up- and downstream emission sources e.g. transport of agricultural inputs and outputs, feed production outside the farmgate, production of inorganic nitrogen (N) fertiliser, processing of products, refrigeration of perishable food. However, these GHG emissions sources are usually of minor importance in MACC assessment.

2.1.1 Enteric fermentation

Most enteric CH_4 emissions can be attributed to ruminant livestock only i.e. cattle, buffalo, sheep, goats, and camels while non-ruminants i.e. swine, horses, mules and poultry have a much lower capacity for CH_4 production. Ogino et al. (2007) stated that enteric CH_4 production can contribute up to 60% of the total GHG emissions in ruminant livestock farms. The total amount of CH_4 is largely dependent on the microbial- composition, activity and population in the ruminal environment which in turn is strongly influenced by factors like feed, feed additives, species and climatic conditions. The internally produced CH_4 is eructated leading to an energy loss of 2-12% of the gross energy intake for the ruminant (Martin et al., 2010). Therefore, it is a desired objective to reduce enteric CH_4 emissions to increase production efficiency. CH_4 and to a lesser extent CO_2 are by-products of bacteria, protozoa and fungi that process the feed components in the rumen under anaerobic conditions to utilisable energy sources for the animal, namely volatile fatty acids like acetates, propionate, and butyrate (Martin et al., 2010). During this so called process of enteric fermentation, certain co-factors are dehydrogenated, thereby leading to hydrogen accumulation in the rumen. The hydrogen surplus is primarily utilised by methanogenic archaea (methanogenesis) for reducing CO_2 to CH_4 (Martin et al., 2010). Methanogenesis is essential for optimal digestion since it reduces the hydrogen content in the rumen, thereby preventing an oversupply of hydrogen and subsequent inhibition of dehydrogenase. While the production of acetate and butyrate lead to a net hydrogen surplus and hence CH_4 emissions, the production of propionate generates a net hydrogen sink and competes with methanogens for the available hydrogen and thereby reduce CH_4 production (Martin et al., 2010).

2.1.2 Agricultural and grassland soils

Croplands and grasslands are major emitters of N_2O and CH_4. Soil N_2O emissions can be either direct or indirect. Direct N_2O emissions are mainly caused by microbial processes of denitrification and nitrification (Eckard et al., 2010). The process of nitrification reduces ammonium (NH_4^+) to nitrate (NO_3^-) in aerobic conditions with N_2O as a by-product, and denitrification is an anaerobic process where nitrate is reduced to nitrogen gas (N_2) that reacts to N_2O (Briner et al., 2012). Both chemical pathways play a significant role in N_2O production since nitrification is essential for transforming N-content of manure, urine or fertiliser into NO_3^- which in turn is used for denitrification. Indirect N_2O emissions are generated by volatilization and subsequent atmospheric deposition of applied N and through leaching and runoff into groundwater (USEPA, 2006). Direct and indirect N_2O emissions require N surplus in the soil caused by N inputs that exceed the N demand of plants. Activities that can add N to the soil include application of N fertiliser, livestock excreta, keeping plant residuals in the field, sewage sludge, cultivation of N-fixing plants, and maintenance of histosols (soil with primarily organic matter content). Excessive fertilisers and/or manure application for promoting plant growth and improving soil conditions are key drivers for soil N_2O emissions, while application before wet soil conditions and soil compaction through animal grazing additionally increases N_2O emissions (Luo et al., 2010).

The process of photosynthesis captures the CO_2 which is then stored as carbon (C) in the plant. If the plant material is not removed from the farmland or grassland, the organic C enters the soil where it is stored. Following exposure to oxygen, the soil organic carbon (SOC) is reduced through microbial decomposition and mineralisation which generates CO_2 as a by-product. The rate of CO_2 emissions is highly variable in different regions depending on the soil type, microbiological ecosystem, C input through plant material, naturally or human induced disturbances and climatic factors e.g. temperature and moisture (Lal, 2004). Agricultural soils have a high C sink potential, but 25% to 75% of the SOC pool in agricultural systems is currently depleted (Lal, 2011). The SOC balance can be disturbed by farming practices e.g. deforestation, tillage and cultivation of organic soils. These can modify C input to soils through decreased plant residuals, increased decomposition rate or erosion of C from arable lands to rivers (Ciais et al., 2010). Another source of CO_2 emission is triggered by fuel consumption during activities like transportation, fertiliser production, fertiliser/manure application and tillage or other field management practices.

2.1.3 Manure storage

Manure storage causes CH_4 and N_2O emissions in livestock systems (Pattey et al., 2005). CH_4 is a product of anaerobic decomposition during manure storage in e.g. ponds, tanks, pits or slurry lagoons. The amount produced is largely determined by factors such as storage duration, climatic conditions of manure storage and manure composition depending on the animal species producing the faeces or urine and the animal diet. A long duration storage and high temperature increases the microbial activities, thereby leading to a higher decomposition rate. Moist conditions favour an anaerobic environment within the storage system and subsequent CH_4 production. Diets with high energy content generate manures with high CH_4 production capacity contrary to low energy diets. A small proportion of the N content in manure and urine is converted into N_2O in aerobic conditions through nitrification. Under anaerobic conditions, N_2O is reduced to N_2 during denitrification (Groffman et al., 2000). Switching from aerobic to anaerobic conditions is likely to occur in solid manure management systems that are exposed to rainfall where pockets with moist (anaerobic) conditions are generated (Eckard et al., 2010).

2.2 GHG inventories and reported GHG emissions from agricultural activities in China and Europe

In the sub-sections 2.2.2 and 2.2.3, data on GHG emissions from agricultural production are presented as reported by Chinese and European GHG inventories which were submitted to the UNFCCC. (EEA, 2014; NCCC, 2012, 2004). Since China published GHG emissions only for the year 1994 and 2005, these sections do not report GHG emissions for the same time periods for Europe and China. GHG inventories are internationally accepted tools to generate GHG emission baselines and are utilised to account for GHG reduction of countries. The quality of the reported data determines the usability of the reports. Although GHG inventories are often based on recommendation by the IPCC (Eggleston et al., 2006), they are subject to limitations. Main limitations arise due to non consideration of consistency, transparency, comparability, completeness and accuracy of inventory reports.

- Consistency relates to the homogeneity of methodologies used to generate inventories in a time series. Application of improved methodologies at later years therefore requires reassessment of older inventories. There is an issue of inconsistency between the Chinese inventories of 1994 and 2005. In both inventories, GHG emission source categories were different e.g. the inventory from 1994 included the emission category "other emissions" which included emissions through grazing and residual burning. Further, emission factors (EF) used to estimate GHG emissions differed between both inventories.
- Transparency implies that methodologies and data sources are fully reported and explained in the GHG inventory report. This allows an independent third party to reconstruct the inventory and correct it if necessary. Although an increasing number of parties report in a transparent manner, there are still countries which do not.
- Comparability between the GHG inventories of different parties is important and ~~thus~~ hence it necessitates the use of similar methods for assessing and reporting GHG emissions. Countries sometimes adopt own EFs, consider a limited set of GHGs, use different timing of inventory generation or apply different Tier methods (Sommer et al., 2004). Although this sharply reduces transparency and comparability of inventories, it still complies with the IPCC guidelines.
- Completeness relates to the coverage of the GHG inventory i.e. consideration of all GHGs source and sink categories, full activity and period coverage. For instance,

some countries only cover a limited set of GHG emissions and thereby partly report GHG emission outputs. Missing to report a complete inventory has negative implications on the emission baseline generation and subsequently on tracking GHG reduction.

- Accuracy is an important issue for the quality of GHG inventories. The IPCC guidelines provide guidance to avoid systematic mistakes but uncertainty of reported GHG output is high. While reporting quality is good in ANNEX I countries as they hold necessary capacities for e.g. assessing statistical data, applying Tier 2 or 3 methods or developing specific EFs, developing countries show higher uncertainties due to lower capacities. However, data certainty is also sector specific. GHG emissions from fuel combustion is a relatively certain estimate compared to GHG output from agricultural activities with high levels of heterogeneity of bio-physical and economic settings, coverage of large areas, spatial and temporal variations of EFs. Annex I countries reported uncertainties of CO_2 from the 'Agriculture, Forestry and Other Land Use' sector between 10 and 100% (Pacala et al., 2010). Uncertainties for CH_4 emissions from rice cultivation and soil N_2O can exceed 50% and CH_4 from enteric fermentation is reported to be 30% and 20% for Tier 1 and Tier 2 methods, respectively (Pacala et al., 2010). Uncertainty in developing countries has even stronger implications as GHG emissions from the agricultural sector usually contribute stronger to total GHG output. However, an increasing number of parties assess the uncertainties of their estimates and allow identifying key uncertainties sources and this may facilitate a reduction of overall uncertainty in future.

Assessing EFs, activity data and subsequent GHG emissions for the agricultural sector is a costly endeavour, depends on ground survey information and self-reporting. Such data is scarce, not independent and often not verifiable by a third party (Pacala et al., 2010). The IPCC guidelines sometimes suggest simple algorithms for assessing national GHG emissions. For instance, multiplying the livestock number with default EFs provided by the IPCC (Tier 1 methodology) delivers estimates only in a low geographical resolution and are temporarily inaccurate. This can lead to less precise quantification of GHG emissions, particularly in the agricultural sector. Using Tier 2 and Tier 3 methods consider also animal size, feeding regime and other production factors and this would facilitate a better understanding on the mitigative effect of mitigation options but requires substantial higher resource input. However, parties that adopt Tier 2 and 3 methods must deliver more detailed data and/or measurements for documentary evidence in order to be accepted for these methods. Inventories associate only N_2O and CH_4 emissions to the agricultural sector which are directly linked to agricultural activities within the farm gate. Some GHG emissions e.g. CO_2 emissions from soils are excluded although these can have a significant contribution to total agricultural GHG emissions. However, C sequestration is covered for the category ‘land use, land use change and forestry’ in the GHG inventory. Upstream emissions e.g. transportation and manufacturing of agricultural inputs and downstream emissions e.g. transportation of agricultural products and fuel consumption occurring during agricultural production are further not accounted for in the category ‘agriculture’. For the latter, GHG emissions are attributed to the category ‘Energy’. Further, GHG emissions occurring outside the national boundaries are not accounted for e.g. in case of imported feed and this has implication also on mitigation efforts. To show the significance of some of these unaccounted GHG emissions, some of these were assessed in chapter 5. GHG emissions attributed to only livestock production are not reported in GHG inventories and can be only roughly abstracted from the GHG output reported for the agricultural sector. Finally, mitigation efforts sometimes cannot be accounted for in GHG inventories and thus these mitigation efforts are invisible. This is particularly important for the agricultural sector with regards to a high heterogeneity within GHG emission sources and with regards to permanence, leakage and additionality. Without accounting for mitigation efforts in GHG inventories, policy makers may have a lower incentive to enforce mitigation policies. Only by improving capacities for measuring, reporting and verifying GHG reduction potentials, particularly in the developing world, GHG reduction from the agricultural sector can be considered (Gerber et al., 2013a). However, uncertainties of reported GHG emissions are a substantial factor for the success of reporting GHG reduction in GHG inventories (Leip, 2010). The IPCC guidance clearly favours the assessment of uncertainties but these should be assessed independently and verified to allow comparison of the GHG inventories. The quality of inputs e.g. EFs and activity data must be improved to allow accounting for GHG reduction at an acceptable degree of certainty (White et al., 2011). Further, GHG inventories need to consider more detailed GHG accountancy methods and thereby consider spatial segregation and temporal variability (Horabik and Nahorski, 2010). Tier 1 methods do not allow for such an assessment and therefore only higher Tier methods should be used. The period of reporting must fit to the temporal variability of GHG emissions to detect changes in e.g. land use and hence the impact of GHG budget of human induced disturbances. Since frequent measurement may not be feasible, detailed, sophisticated and standardised models needs to be approved and applied for Tier 2 and Tier 3 methodologies. With regards to soil

C, this could allow accurate assessment of stock changes. For enteric GHG emissions the link between feed quality and enteric emissions must be improved (Hristov et al., 2013). This could allow accounting for GHG reduction through feed supplements but requires a detailed assessment of baseline feeding regimes and other production factors and reassessment of baselines from older GHG inventories. Further research is required to assess the relationship between dietary nutrients and enteric GHG emissions. CH_4 prediction equations could be used but these can show low prediction accuracy compared to experimental data (Histrov et al., 2013). Further, the accountancy boundaries of the national GHG inventories would not allow accounting for emission reduction of imported feed compounds for the importing country. A solution could be consumption-based GHG accounting but this requires an international agreement on alternative GHG reporting. Finally, the IPCC guidelines should include a set of mitigation options that allow assessment of GHG reduction potential based on a combination of baseline crop, soil, animal species, climate and management. However, the development of GHG inventories is progressing and in future it might be possible to report for mitigation efforts particularly in the agricultural sector.

2.2.1 Emission intensities versus absolute emissions

China started the debate on using emission intensity i.e. GHG emissions per unit of product as an alternative metric for absolute GHG emissions. Assessing emission intensity has several advantages. First, production efficiency e.g. output per animal can be directly linked to the emission output. Second, this metric allows a direct comparison of GHG emission per production output within commodities and also between commodities if a common measure e.g. unit of GDP is used. Third, it can indicate cost-efficiency of mitigation as increased production efficiency might increase producer profits by simultaneously reducing GHG output and this indicates lower cost for mitigation efforts. Fourth, this metric can accommodate emission reduction despite increasing production levels (Gerber et al., 2013b). Fifth, the assessment of reduction potentials with emission intensity is likely to minimise the tradeoffs between mitigation, food security and producer welfare (Gerber et al., 2013b). Finally, emission intensities do not falsely report GHG reduction due to leakage e.g. total GHG output of a country can decrease if production is relocated outside accounting boundaries. This metric poses a major disadvantage as absolute GHG emission output is neglected but the amplitude of climate change is triggered by total GHG emissions in the atmosphere and successful mitigation requires GHG reduction in absolute terms. Although emission intensities will decrease in the livestock production globally, the increasing production levels will cause higher GHG output in future (Gerber et al., 2013a). This is particularly evident in China with a strongly increasing livestock production. Further, the Kyoto protocol and most other international agreements proposed absolute GHG emission caps and decrease of emission intensity is not applicable to these targets. Therefore, GHG inventories report absolute GHG emissions. However, there is a debate on whether to enforce intensity-based caps, absolute caps or a mix of both to address global climate change mitigation but this is subject to international negotiation.

For the livestock sector which utilises imported commodities outside accountancy boundaries, the advantage of using emission intensity metrics is that the consumer can make dietary choices to reduce resource use. This could further trigger consumer induced mitigation as an alternative to governmental imposed mitigation (Gerber et al., 2013a). This requires adequate labelling and certification programmes to inform the consumers and standardised metrics and methods to assess the related carbon intensities (Gerber et al., 2013a). Further, life cycle analyses (LCA) are required that estimate GHG emissions ideally throughout the product's life cycle following standardised guidelines. However, this is a time consuming task and can be sometimes problematic. Finding a common unit e.g. kg meat can be difficult as the livestock sector produces different commodities and even specialised livestock farms produce by-products that are different from the main product e.g. beef as a by-product in milk production. In case of milk production, this common unit can be further segregated into energy corrected milk and these units are not directly comparable (Yan et al., 2011). In this case the inputs and outputs should be separated between production of milk and beef but looking at both commodities separately may not be beneficial for addressing mitigation possibilities (Yan et al., 2011). Although ideally LCAs are boundary less, in Europe most of the LCA studies for milk production focus on "cradle to farm gate" due to lack of data and thereby fail to assess the entire life cycle of the products (Yan et al., 2011). Using emission intensity can be generally applied to GHG inventories and also MACCs. However, since MACCs are a policy tool for assessing cost-efficiency of achieving binding GHG reduction targets it is favourable to use the metric of absolute GHG reduction as this can be potentially accounted for in GHG inventories.

2.2.2 Agricultural GHG emissions in China

In 2005, agricultural activities emitted 820 Megaton (Mt) CO_2e in China which is 11% of total GHG emissions. In 1994, total agricultural GHG emissions were 605Mt CO_2e and contributed to 15% of total Chinese GHG emissions (Table 2.1). In 1994 and 2005, enteric CH_4 emissions have been the largest source of agricultural GHG emissions i.e. 35.3% and 36.8% of total agricultural emissions, respectively (Table 2.1). Cattle (dairy and non-dairy), buffalos, goats, sheep and pigs were the key sources of enteric CH_4 emissions in these years. Although pigs are non-ruminants, the enormous pig count in China leads to high total GHG emissions from enteric fermentation. Livestock manure management was the fourth largest source with GHG emissions of 31.8 and 143.8 Mt CO_2e in 1994 and 2005, respectively (Table 2.1).

Table 2.1: GHG emissions from different sources in China.

Year	Agriculture - total†	Agricultural soils*	Enteric fermentation+	Manure management†	Rice cultivation†	Others (grazing, residue burning, etc.)*
1994	605	195	214	32	129	36
2005	820	208	302	144	167	

† Includes N_2O and CH_4 emissions
* Includes N_2O emissions
+ Includes CH_4 emissions
Source: altered from NCCC, 2004 and 2012

2.2.3 Agricultural GHG emissions in the EU-15

In 2012, the agricultural sector in EU-15 emitted 373 Mt CO_2e which is 10% of total GHG emissions in EU-15 (Table 2.2). Compared to 2005, there has been a reduction of 5% (21 Mt CO_2e), respectively (Figure 2.1).

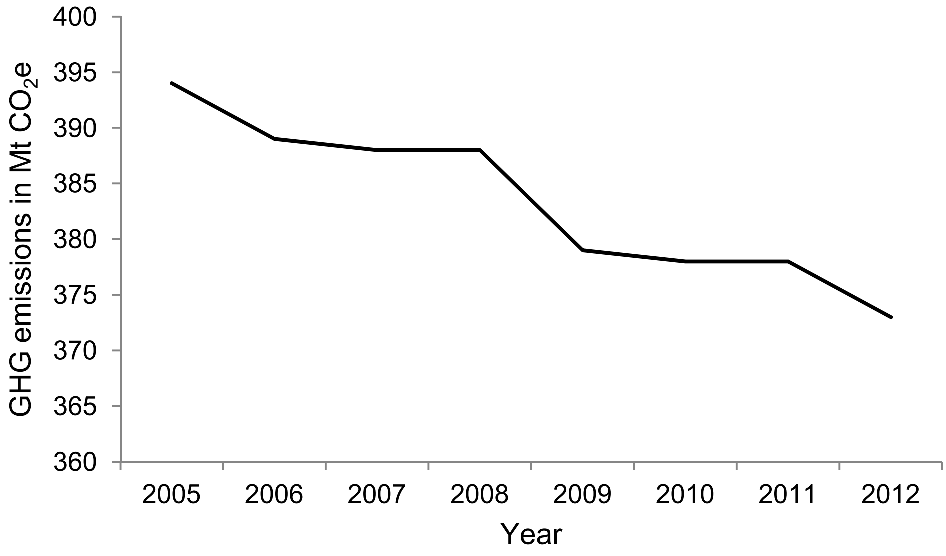

Figure 2.1: Agricultural GHG emissions in EU-15 between 2005 and 2012.
Source: altered from EEA, 2014

In 1990 and 2012, agricultural soil was the largest GHG emission source of total agricultural GHG emissions in EU-15 i.e. 52% and 50% of total, respectively (Table 2.2). Enteric fermentation was the second largest source i.e. 32% of total in both years (Table 2.2). In both years, cattle (dairy and non-dairy) were the primary emitters within this GHG emission category. Manure management was the third largest source of GHG emissions with 69 and 61 Mt CO_2e in 1990 and 2012, respectively (Table 2.2).

Table 2.2: GHG emissions from different sources in EU-15.

Year	Agriculture total†	Agricultural soil*	Enteric fermentation⁺			Manure management†
			Total	Dairy cattle	Non-dairy cattle	
1990	443	229	142	57	61	69
2012	373	188	121	45	55	61

* Includes N_2O emissions
⁺ Includes CH_4 emissions
† Includes N_2O and CH_4 emissions
Source: altered from EEA, 2014

2.3 Mitigation options for livestock production

Based on the identification of GHG emission sources from livestock production, it is possible to identify mitigation options that target each GHG emission sources specifically. There are vast mitigation options available for reducing GHG emissions from livestock production. The purpose of this section is not to review the whole literature on mitigation possibilities for the livestock sector. This section intends to give a brief description of the supply-side mitigation options used throughout this research study. However, to illustrate this scientific field and recent developments, alternative mitigation options are discussed for each GHG emission source based on the review of Hristov et al. (2013) and supplemented by other sources as indicated. Special emphasis is given to the dietary mitigation options as these are of high importance for the MACCs developed in this thesis and with enteric fermentation being the largest GHG emission source also for mitigation activities. The criteria for selecting mitigation options that were used for the MACCs are described in section 3.3.2.1. For the MACC development it is most significant to work with reliable data related to activity levels, mitigation potentials, baseline projections and economic data. This holds true particularly for large regions like China and EU-15. Therefore, mitigation options were selected where necessary data is available. However, these MACCs can be updated if new and reliable information is available.

2.3.1 Enteric fermentation

2.3.1.1 Dietary lipids

Lipids as feed supplements can reduce the production of hydrogen and consequently enteric CH_4 emissions through several mechanisms i.e. inhibition of rumen protozoa, biohydrogenation, fibre digestion, supression of methanogens and lowering dry matter intake (DMI; Eckard et al., 2010). The structure of the lipids e.g. polyunsaturated, monosaturated, saturated or medium chain fatty acids is crucial in determining the mechanism for CH_4 reduction and hence the total CH_4 reduction potential (Martin et al., 2010). Although medium chain fatty acids show high GHG reduction potentials, such lipids can have adverse effects on human health, whereas polyunsaturated fatty acids could be beneficial for human health (Martin et al., 2010). The lipid content of the DMI should not exceed 6-7% since higher concentration may reduce the digestibility and voluntary feed intake (Beauchemin et al., 2008). This threshold is of particular importance for lipid supplementation in diets that contain fats in the concentrate compound. Using lipids from by-products of industrial food processing e.g. cottonseed, brewers grains or maize meal can be an appropriate source of lipids for large scale implementation. For grazing animals, an alternative could deliver gene manipulation of grasses to accumulate fatty acids or adding fatty acids into drinking water of the animal (Grainger and Beauchemin, 2011).

Meta-analyses have shown the effect of lipid supplementation. Eugène et al. (2008) showed under consideration of 25 diets (from 7 studies) that DMI was reduced by 6.4% of lactating dairy cows but gross energy intake was not affected as the lipid supplementation compensated the reduced DMI of cows. This analysis also shows that daily methane production was lower i.e. by 8% (g/day) but this was mainly explained by the decreased DMI. Expressed as gross energy intake or digestible energy intake CH_4 was reduced by 9%. The study concluded that lipid supplementation improved feed efficiency and thus decreased indirectly CH_4 production but the inhibitory effect on CH_4 production is not the main reduction source. The meta-analysis of Patra (2013) considered 29 experiments and showed that fat supplementation decreased methane production linearly expressed as g/day, g/kg DMI, g/kg digestible DMI, g/kg milk and % of gross energy intake if diets contained more than 5% fat. In case of fat concentrations below 5% g/day or g/kg digestible DMI enteric CH_4 emissions were not significantly affected, while the expression in g CH_4/kg milk production showed a significant and linear reduction. This study also reported a negatively affected digestibility of DMI by fat supplementation due to decreased number of protozoa and lower activity of various bacteria and degrading in enzymes in the rumen. Milk production was affected positively by fat intake. However, results on effect of dietary lipids on animal productivity are inconsistent. Due to the complex response of lipid supplementation there is a need to generate standardized guidelines for lipid supplementation to ensure CH_4 reduction and at least no decline in productivity (Grainger and Beauchemin, 2011). Further research must investigate these complex mechanisms to provide more homogenous evidence for the reaction of lipid supplementation. Further, there have been issues regarding the longevity of CH_4 reduction effect of lipid supplementation. Woodward et al. (2006) showed that vegetable and fish-oils significantly decreased CH_4 emissions in short-term but after 11 weeks of supplementation this effect was not observed. They could not prove an increased milk production after lipid supplementation. Lipids are likely to reduce CH_4 emissions but can decrease feed intake and therefore also productivity but this depends on the feed composition of the animal (Hristov et al., 2013). However, Grainger and

Beauchemin (2011) concluded that lipid supplementation can increase animal's growth or yield in case of increased the energy availability for the animal.

2.3.1.2 Dietary probiotics

Probiotics are microorganisms that can modify the ruminal ecosystem. Active microbiological yeast cultures based on *Saccharomyces cerevisiae* are widely applied in dairy farms in the United States of America (USA) and Europe for improving production efficiency (Robinson and Erasmus, 2009). Yeast products can show positive effects in nutrient utilisation, rumen fermentation characteristics, milk production and daily gain (Patra, 2012). The meta-analysis of Desnoyers et al. (2009) stated that 157 experiments on milk production by supplementation of *Saccharomyces cerevisiae* showed strongly inconsistent results. This study showed that yeast supplementation increased average milk yield by 1.2g/kg of body weight, fat content by 0.05% and DMI intake by 0.44g/kg of body weight. However, this meta-analysis showed no effect on milk protein content. The meta-analysis of Poppy et al. (2012) of only peer-reviewed publications showed a mean difference between cattle supplied with *Saccharomyces cerevisiae* and untreated cattle of 1.18 kg/day, 1.61 kg/day, 1.65 kg/day, 0.06 kg/day and 0.04 kg/day for milk yield, 3.5% fat corrected milk, energy corrected milk, milk fat yield and milk protein yield, respectively. However, they also stated that the heterogeneity of reported results was substantially. This can be due to influencing factors as dose, type of diets, strains of yeast, physiological stage and feeding systems (Patra, 2012). Such yeast strains could be selected that reduce CH_4 production while improving rumen fermentation, fibre digestion and energy availability for the animal (Newbold and Rode, 2006). These microorganisms divert hydrogen from methanogenesis to acetogenesis in the rumen and thereby increases production of acetate and reduces CH_4 production (Moran et al., 2008). Many studies reported a decreasing effect of *Saccharomyces cerevisiae* and other microbial feed additives on enteric CH_4 production but a large share of studies examined this effect only *in vitro* (Patra, 2012). More research is required to understand the interaction between type of diets and microbial feed additives and thus the CH_4 reduction potential *in vivo.*

2.3.1.3 Dietary nitrate

Nitrate in the rumen obstructs the chemical pathway that leads to generation of CH_4. Since nitrate is rich in oxygen, the oxygen is utilised in anaerobic systems like the rumen and thereby replaced by hydrogen that is not available for CH_4 production (Lee and Beauchemin, 2014). An overdose can lead to health problems for the animal. Therefore, nitrate level in the basal diet has to also be considered and nitrate supplementation in protein rich diet is not recommended on health grounds. In this case, the diet should be only supplemented with a low dose of nitrate. It is important to gradually adapt the animal to nitrate supplementation to prevent nitrite toxicity for the animal (Leng, 2008). In case of increasing total N content in the diet after nitrate supplementation, N excretion can increase. However, this effect is not expected if there is no change of N content in the diet (Hristov et al., 2013). Nitrate supplementation shows an impressive enteric CH_4 reduction. No meta-analyses for the effect of nitrate on enteric CH_4 were found and therefore reduction potentials of nitrate supplementation are shown as reported in following studies. Hulshof et al. (2015) showed that methane emissions were reduced by 32% when feeding steers with nitrate (85g/day) instead of urea (125g/day). Expressed as emission per kg DMI and gross energy intake, methane reduction was reduced by 27% and 28%, respectively. Guyader et al. (2015) showed that addition of nitrate reduced 22% CH_4 g/kg DMI which corresponds to a 9.8% reduction per percentage unit of nitrate fed. This result is line with other studies that reported a percentage reduction of nitrate fed in the range of 7.9% and 12.2% (Guyader et al., 2015). This allows the conclusion that nitrate addition has a similar effect on CH_4 reduction regardless of the diet or animal species. This study showed further that the combination of linseed oil and nitrate has an additive affect on mitigation and did not show changes in digestibility; hence is recommended as an efficient mitigation option. Due to toxic side effects it is required to precisely add this additive to the animals and monitor the health of the animal. A possibility to control the nitrate intake could be the use of slow-release encapsulated nitrate. Research must focus on the adaptation of the ruminal ecosystem to nitrate feeding and understand the effect in long-term studies. Further, potential increase of NH_3 emissions must be understood. There is a need for more *in vivo* studies to understand the GHG emission reduction potential of nitrate supplements at farm-scale and further research is required to prove no residuals of nitrate in the product of the animal fed with nitrates (Hristov et al., 2013).

2.3.1.4 Dietary saponins and tannins

Saponins and condensed tannins (CT) are secondary plant compounds that can be found in varying concentrations in many plants. Patra (2012) reported methane inhibition of tea saponins between no effect and 15.5% and for tannins between no effect and 32.6% depending on agent used. Tannins have been widely researched as a promising mitigation option to reduce enteric CH_4 emissions. Eckard et al. (2010) stated that tannins can be used as feed additives for inhibiting the activity of methanogens, improving animal productivity by increasing the digestion efficiency of amino acids, decreasing N content in urinary excretion and increasing N content in the faeces (Eckard et al., 2010). The effect on these factors is however largely inconsistent throughout literature (Hristov et al., 2013) and this can be attributed to the type, concentration and protein binding capacity of tannins. Further reasons for inconsistencies are variable techniques to measure tannin concentration, failure to distinguish between condensed and hydrolysable tannins and the level of intake for optimal supplementation. The meta-analysis of Jayanegara et al. (2011) including 30 experiments with 171 treatments of *in vitro* and *in vivo* experiments stated a CH_4 reduction with increasing tannin levels in the feed. For the *in vitro* studies the response was quadratic and the *in vivo* studies showed a linear decline. This study reported strongly varying results of CH_4/DMI at low level tannin supplementation and this might explain heterogenic results in previous studies but with increasing levels of tannins supplementation variation decreased. They concluded that CH_4 reduction is associated with a lower digestibility of nutrients. However, this meta-analysis did not show that reduced CH_4 production is attributed to a decrease in the acetate-to-propionate ratio but CH_4 was reduced per unit of digestible organic matter which could imply the direct inhibitory effect of tannins on CH_4 production. Carulla et al. (2005) reported that feeding extracts from *Acacia mearnsii* to sheep did not reduce fibre digestibility while reducing CH_4 production by 12%. Tannin can be fed in extracted form e.g. from chestnut wood, which is relatively cheaper as compared to extraction from other plants. Alternatively, they can be fed as tannin containing plants. CTs can be found in a large pool of plants, mainly tropical shrub legumes, but these plants may lack the agronomic feasibility to replace traditional forage sources (Beauchemin et al., 2008). The meta-analysis of Archimède et al. (2011) showed that CH_4 production was reduced by 29% if animals were fed with high tannin legumes compared with those fed with low tannin legumes. As high CT concentrations in the diet can reduce the digestibility and voluntary feed intake of the animal, this additive should be fed in a controlled environment e.g. within a housing system to avoid productivity losses or if supplemented through forage, animal's health must be monitored

(Beauchemin et al., 2008). Research should focus on the long-term effects on CH_4 reduction of tannins and particularly on identifying tannins that do not compromise animal production. In order to recommend tanniferous forages as a mitigation option, research must focus on the CH_4 reduction potential and also consider the agronomic characteristics as tannin rich legumes can be limited by this factor. Further, the effect of tannin supplementation on manure and N_2O emissions must be understood as only limited evidence is available. Hao et al. (2011) showed that application of *Acacia mearnsii* at 25 g/kg dry matter (DM) had no impact on CH_4 and N_2O emissions from manure but this could be attributed to the low level of tannin supplementation.

Saponins are available in high concentrated form in e.g. quillaja, yucca and tea. Saponins in tea plants are particularly important for China as these can be widely available from the by-product of the massive tea production industry. However, literature is inconsistent on the CH_4 reduction potential of saponins. The meta-analysis of Jayanegara et al. (2014) including 23 *in vitro* studies showed that increasing levels of saponin decreased CH_4 emission linearly. However, when added above 500mg/g DM saponin had little effectiveness on further CH_4 reduction. Exceeding this threshold also did not show adverse impact on the animal productivity and in this regard saponins are advantageous over tannins. This study showed that acecate proportion decreased and propionate proportion increased linearly with increasing levels of saponin. After adding saponins, CH_4 emissions could be reduced by a lower methanogen population and reduced activity of methanogenes. However, this could not be accessed through the meta-analysis due to insufficient data availability. There are currently only few *in vivo* feeding trails for saponin supplementation available. Holtshausen et al. (2009) could not show an effect on CH_4 production, rumen fermentation, digestibility and milk production by supplementing saponin at a dose of 10g/kg DM in early lactating cows. This is in line with Li and Powers (2012) that also could not show significant effect of saponin supplementation on CH_4 reduction of ruminants. Contrary to this the study of Mao et al. (2010) showed significantly reduced CH_4 emissions of growing lambs by 27%/kg DMI. *In vivo* experiments show inconsistent results on the mitigative effect of saponin addition. One reason could be that different types of saponin show different anti-protozoal properties and thus the effect on CH_4 reduction varies. For instance, it was shown that yucca saponins reduced CH_4 emissions more strongly than tea saponins (Jayanegara, et al., 2014). Further research is required in order to understand the exact mechanism of different saponin structures on CH_4 production. However, this remains a demanding task due to the large structural diversity of the substances even within single plant species (Jayanegara et al., 2014). The effect of the diet on the GHG reduction potential must also be investigated further. There is also an issue with adaptation of rumen population to saponin supplementation and this is a challenge for this mitigation option (Patras, 2012).

2.3.1.5 Genetic improvement

Improving the genetic merit for increased yield, fertility or survival is widely used in livestock production and can reduce GHG emission per unit of product effectively (Bell et al., 2011; O'Brien et al., 2010; Wall et al., 2010). However, depending on the livestock system, the targets of breeding activities are different. For instance, in some member countries of EU-15, productivity of animals is already high and hence breeding activities could be prioritised that focus on e.g. fertility traits. In chapter 4, genetic selection therefore focuses on improving productivity. Higher productivity of animals leads to higher efficiency (feed conversion rate) and maintenance of the production level with decreased number of animals which thereby reduces total GHG emissions. Chapters 5 and 6 focus on animal selection for reduced enteric CH_4 production and higher milk yield (Wall et al., 2010). Different groups of cattle showed persistent CH_4 output variations between 40% and 55% while being fed with the same diet (Goopy et al., 2006), and genetic traits have a high heritability, thereby allowing to achieve reduction potentials in mid-term future (Haas et al., 2011). However, selection of this trait is limited by high expenses in measuring animals' CH_4 output (Hegarty and McEwan, 2010). A recent solution is to select animals based on proxy traits for CH_4 reduction (Haas et al., 2011).

2.3.1.6 Alternative mitigation options

Additives classified as inhibitor for the activity of rumen archea include also bromochloromethane and chloroform. These inhibitors showed *in vivo* CH_4 reduction potentials of up to 50% for sheep, goat and cattle but science must prove long-term effects of CH_4 reduction and consequences for the productivity of the animal. However, bromochloromethane is currently banned and thus cannot be recommended as a mitigation option. Chloroform is categorised as a carcinogen and thus public acceptance for this additive might be low. Besides nitrate, there are other electron receptors (H_2 sinks) available including sulphates, fumarate and nitroethane. Fumerate have not shown to reduce CH_4 production throughout literature but leads to a decreased feed intake and thereby reduces animal productivity and increases emission intensity. However, nitrate might be the most promising agent in this category and was therefore considered for the European MACC. Antibiotics as ionophores can increase the productivity of livestock and thus decrease emission intensity per output. This agent showed a reduction of CH_4 production by up to 18% (Patra, 2012). Monesin is probably the most studied amongst the antibiotics. However, antibiotics as feed supplement have been banned in the EU and impose health risk for the consumer; thus cannot be recommended as a mitigation strategy. Although in China antibiotics are largely applied in livestock production, it is expected that antibiotics will be also banned in future. Alternatives to tannins and tea saponins are essential oils. Although there is a large *in vitro* evidence base, there are only few studies *in vivo*. Therefore, it is difficult to judge on the abatement potential of this mitigation option in practical application. Histrov et al. (2013) showed that leaves from *Origanum vulgare* can reduce CH_4 production significantly with also an increased feed efficiency and milk production. But this has not been assessed in long-term studies. Exogenous enzymes could also deliver a good mitigation option as they increase feed efficiency and thus decrease emission intensity per product. However, strong inconsistencies in the experimental data do not allow recommending this mitigation option. Defaunation is well discussed in the scientific literature and average CH_4 reduction of 10% were shown but results showed a strong variability (Morgavi et al., 2010). The high uncertainty and variability of the CH_4 reduction effect and the difficulty to maintain the altered fauna does not allow recommending this mitigation option so far. Vaccination to suppress the activity of rumen archaea and bacteria were proven to successfully reduce CH_4 production *in vitro* but practical application remains difficult. Sequencing the genome of *Methanobrevibacter ruminantium* which is a highly potential methanogen allows for new vaccination for CH_4 reduction. The successful vaccination of mother animals has shown that their kids also maintained a lower CH_4 production (Abecia et al., 2011). This finding shows the potential of large-scale and long-term application but practicability of the vaccines must be improved. However, it remains questionable if this feed supplement will be accepted by the consumer. Increasing the concentrate share in the feed compound can reduce CH_4 production and also reduces GHG intensity by increased production output (Beauchemin et al., 2008; Martin et al., 2010). However, in intensified livestock production systems in Europe and China, feeding concentrate is common practise and exceeding a concentrate share of 50% implies negative consequences for the animal (Beauchemin et al., 2008). Further, GHG emissions from producing concentrate is an issue and it needs to be clarified that the GHG reducing effect is positive while considering GHG emissions outside the farm gate. Science also focussed on advancement in feed management including feed processing and precision feeding. Reducing the particle size of the forage leads to increased digestibility, reduced energy consumption for digestion, improved feed intake and productivity. This thereby decreases the emission intensity per product. Precision feeding describes feeding to match the animal's nutrient requirement and thereby improve animal health, productivity and minimise nutrient excretion. However, this technology to decrease emission intensity could be rather costly as it requires investments in infrastructure and sufficient technologies. Finally, science elaborated also advanced housing for the animals. Biofiltration of mechanically ventilated air can capture the GHG emissions released from the housing system but is likely to be achievable only at high costs (Montes et al., 2013). Very few studies assessed the effectiveness of biofiltration and further research is required. However, housing systems with low CH_4 concentration would require large and effective biofiltration systems, probably at high costs. Despite genetic improvement of the livestock for increased productivity and decreased enteric CH_4 output, research recently focussed on selection of traits for improved health, fertility and reduced mortality. These traits improve the ratio between production output and GHG emissions i.e. reduce emission intensity by increasing herd's productivity. In case of improved fertility, selection of these traits allows to maintain a lower number of animals for replacement purposes. Fertility of animals is particularly important for the dairy industry as fertility decreased over the last decades by focussing only on higher producing animals.

2.3.2 Agricultural soils

2.3.2.1 Reduced grazing intensity

Depending on the grassland condition, it can be beneficial to increase or decrease the grazing intensity and thereby improving the net primary production (NPP), SOC sequestration, soil conditions and the mix of plant species. In Chinese grasslands, a reduced grazing intensity may improve the grassland conditions as degraded soils with low NPP and high grazing intensities are predominant. Thereby, the NPP may increase while grass utilisation by the livestock decreases and this leads to subsequently higher amounts of plant residual matter that will be sequestered in the soil and thereby increasing the soil C storage. However, a reduced grazing intensity can also increase utilisation rate of the grass depending on the previous stocking and grazing intensity. This must be considered for selecting appropriate grazing intensities to increase soil C. Grazing intensity can be reduced at different levels as for instance to medium grazing intensity (MGI), light grazing intensity (LGI) and grazing prohibition and each of these intensities will differentially affect NPP and grass utilisation (Patton et al., 2007).

2.3.2.2 Reduced and no tillage

Tillage practices on the field lead to physical breakdown of residues and exposes soil organic matter to oxygen that in turn enhances the activity of aerobic microorganisms. There is also a subsequent increase in decomposition and mineralisation of soil organic matter, thereby increasing CO_2 emissions from soil (Paustian et al., 2000). To avoid these impacts from conventional tillage, different soil management techniques are available, including reduced tillage and no-tillage. Reduced tillage includes management techniques as ridge tillage and shallow ploughing. It disturbs the soil to a lesser extent compared to conventional tillage and can increase soil C storage if it replaces conventional tillage (Abdalla et al., 2013; Vleeshouwers and Verhagen, 2002). No-tillage refers to a practice in which the soil is not deeply loosened for seed application and hence major share of the soil surface is covered by plant residues (Soane et al., 2012). No-tillage practices can reverse the process induced by tillage practices and thus increases soil C storage (Ogle et al., 2005). The effect on C sequestration and soil N_2O emissions for these management techniques varies considerably, depending on factors such as soil texture and climate e.g. temperature and moisture (Lal, 2004). No-tillage practices can show higher soil N_2O emissions due to increased anaerobicity, water retention and compaction of the soil (Smith et al., 2001). However, N_2O fluxes of no-tillage practices may change over time and can become lower over an extended time period as compared to conventional tillage due to the improving soil conditions (Six et al., 2004). The meta-analysis of Kessel et al. (2013) including 41 studies globally showed that reduced and no tillage did not change average N_2O emissions significantly. However, separating by climate, humid climate showed no significant differences in average but dry climate showed differences with increased N_2O emissions by 38% in short-term and decreased N_2O emission by 34% in long-term (>10 years). Reduced tillage or no-tillage practices may lead to growing weed populations which requires proper weed management with e.g. herbicides and can lower the crop yield (Locke et al., 2002). Kessel et al. (2013) showed that reduced and no tillage decreased crop yield in most of the studies compared to conventional tillage with dry climates showing higher yield losses than humid climates. Further, the long-term effect on yield recovery was low and did not recover the yield under conventional tillage practices. GHG savings can arise from reduced fossil fuel use from no ploughing or ploughing in a reduced depth and reduced fertiliser usage (Holland, 2004). However, the permanence of soil C storage is an important issue as one-time tillage can significantly decrease the soil C storage. For humid temperate climatic conditions in Europe, reduced tillage is a more appropriate technique as compared to no tillage, since it allows faster soil warming in spring, better control of perennial weeds, rapid N mineralisation and crop growth (Mäder and Berner, 2012). However, Powlson et al. (2014) claims that the United Nations Environment Programme overestimated the mitigation potential of reduced and no tillage practices as the additional organic carbon in soil is relatively small and GHG reduction through reduced fuel combustion might be offseted by possible increase in N_2O emissions. Therefore, this study concludes that reduced GHG emissions are small and the ancillary benefits have more importance for the agricultural sector than the mitigative effects.

2.3.2.3 Cover crops

Cover crops, catch crops or green manure are planted during the fallow period of annual cash crops to provide a temporary vegetative cover. This helps to reduce N_2O emissions and N leaching, depending on the N uptake abilities of the cover crop species (Dabney et al., 2007). There is the issue of increased N_2O emissions from soils if cover crops are ploughed into the soil. The meta-analysis of Basche et al. (2014) including 26 peer-reviewed publications stated that total N_2O emissions were significantly higher when cover crops residues were ploughed into the soil and areas with higher total precipitation and variability in precipitation tend to increase total N_2O emissions. They concluded that non-legume species have a higher N_2O reduction potential but more research is required to fully understand the impact of cover crops on global N_2O emissions. Ancillary benefits of cover crops are prevention of soil erosion, weed control, protection of water bodies if N leaching is reduced, improvement of soil quality and fertility and reduces risks of diseases and pests (Montes et al., 2013; Snapp et al., 2005). Some cover crop species may also improve nutrition availability of cash crops e.g. by delivering an additional N source for the cash crop if they are not removed from the field. Soil C storage can also be increased if the additional plant material is left on the field or ploughed into the soil. The meta-analysis of Poeplau and Don (2015) including 139 plants at 37 different sides stated that cover crops increased soil C by 0.32 ± 0.08 tonne per hectare and year and could to lead to an average maximum increase of 16.7 tonne per hectare. However, C sequestration rate is vulnerable to changes, and the soil shows high varieties in sequestration activity (see section 2.3.2.2). Cover crops have no significant effect on biomass production of the cash crop (Constantin et al., 2010).

2.3.2.4 Reduced fertilisation

Application of N fertiliser leads to N-leaching, N_2O, NH_3, and CO_2 emissions (latter one being associated with fertiliser production, transport, and application). The amount of direct and indirect N emissions is determined by the N input and also by various other factors, including soil C content, soil texture class, soil drainage, fertiliser type, crop type and soil pH (Bouwman et al., 2002). Over-fertilisation is a common problem in the croplands of China and Europe. In EU-15 for instance, the average N-surplus (i.e. N input to the soil that is not utilised by plants and therefore volatilised via leaching or atmospheric N deposition) was 56 Kg N/ha in 2008 with spatially large differences (EC, 2014a). This adverse situation is highlighted by the large share of European croplands being identified as Nitrate Vulnerable Zones[1] (NVZs). Fertiliser application is restricted in NVZs to reduce the N pollution by e.g. fertilisation only in dry months, reduced fertiliser use and season restricted manure application (Oenema et al., 2009). There are mitigation options available as improved timing of fertilisation, improved frequency of application, using enhanced-efficiency N fertiliser to ensure an efficient N fertiliser application and thereby could lead to a reduction of direct and indirect N emissions, environmental and water damages (Brink et al., 2005). Often an adequate measure to reduce negative externalities of fertiliser application is simply the reduction of fertiliser input, ideally to such an amount that the N fertiliser input equals the N demand of the plants.

2.3.2.5 Optimised fertiliser application

Besides optimising the amount of N fertiliser input, techniques like timing of fertilisation or split fertiliser application also have significant influences on soil N_2O emissions. The first technique restricts application of fertilisers e.g. during autumn or winter as farmers

[1] The surface area of NVZs of the EU member states is: Austria 100%, Belgium 61%, Bulgaria 0%, Czech 38%, Germany 100%, Denmark 100%, Estonia 7%, Spain 21%, Finland 100%, France 53%, Greece 19%, Hungary 45%, Ireland 99%, Italy 27%, Lithuania 100%, Luxembourg 100%, Latvia 13%, Netherlands 100%, Poland 2%, Portugal 10%, Romania 0%, Sweden 49%, Slovenia 100%, Slovakia 38%, United Kingdom 81% (Oenema et al., 2009).

sometimes apply their manure to the fields to increase the storage capacities or to increase the pasture growth in the winter. In temperate zones like in Europe, these seasons have a high precipitation rate that leads to wet soils and thereby indirect soil N_2O emissions. Additionally, during this period, there are usually none or limited cash crop activities in Europe and hence a strongly reduced N demand by plants. However, the effectiveness of this mitigation option is dependent on regional climatic conditions. Split fertilisation as compared to single fertiliser application can also reduce soil N_2O emissions. Fertiliser application can be done several times a year in such a way that plants receive the fertiliser during their growth period with highest N requirement and thereby reducing the risk of N loss. A lower N-content in the soil by applying a smaller amount several times a year compared to a large amount applied one time per year subsequently leads to lower N losses as in latter case the N excess in the soil causes direct and indirect N_2O emissions. On an overall, mitigation options that maximise N use efficiency minimise N losses to the environment (Burton et al., 2008).

2.3.2.6 Nitrification inhibitors

Nitrification inhibitors have been researched over the last 5 to 6 decades. Nitrification inhibitors can reduce N losses from N fertilisation by inhibiting the N transformation process during nitrification such that the N remains in a more immobile form in the soil i.e. NH_4^+ (Luo et al., 2010). Nitrification inhibitors can be directly applied with N fertiliser or as a spray after N fertiliser application. Nitrification inhibitors such as Dicyandiamide (DCD) can also increase the dry matter (DM) production of crops or grasslands. However, the results are largely inconsistent due to experiments in different climates and soil types (Di and Cameron, 2005; Monaghan et al., 2009). Although nitrification inhibitors show significant reductions of direct and indirect N losses, residuals of these chemicals in plants or animals being fed with these plants can be an issue for human health. The nitrification inhibitor *dicyandiamide* is one nitrification inhibitor and has drawn a lot of attention in science. However, *dicyandiamide* was banned recently in New Zealand due to residuals in the plants which can be also the case for other type of inhibitors.

2.3.2.7 Alternative mitigation options

During the last decade enhanced efficiency nitrogen fertilisers which also can include nitrification inhibitors have been studied. These allow an increased N plant uptake and also reducing nitrogen losses to the environment (Snyder et al., 2014). While in New Zealand total N uptake was increased by >30%, N_2O emissions and nitrate leaching were reduced by >30% and >80%, respectively (Snyder et al., 2014). Meanwhile, ammonia emissions increased. Biochar is also focus of science and application of 15-30t/ha could reduce N_2O emissions by over 37% (Snyder et al., 2014). Despite strong scientific efforts, this mitigation option is not well understood and thus requires more research. It might also be a high cost measure considering the production, transport and application of the biochar. There have been strong research efforts in Europe recently to understand how to improve species compositions in pasture. One strategy could be to introduce plants with an increased N-use efficiency to reduce N input requirements and thus reduce nitrogen loss to the environment. Further, improved crop rotation with legumes could deliver a high mitigation potential (Bryan et al., 2011). This mitigation option could lead to yield losses in short term as the cropping intensity is reduced. However, it is expected that in medium- and long-term an increased crop yield is expected due to improved soil fertility and water holding capacity. Since long time there is a debate on plant breeding for increasing yield. Selection of traits for a higher adaptability to changing climate conditions could maintain yield levels. Finally, there are different grazing management techniques available as these shown above. Such management options can range from cutting management and improved grazing in terms of rotational grazing to increase the productivity of the pastures and lower soil disturbances, thereby increasing carbon sequestration.

2.3.3 Manure storage

2.3.3.1 Anaerobic digestion

Anaerobic digestion of manure in closed vessels converts organic material to biogas and can be used as substitute for fossil fuels for generating heat or electricity (Hristov et al., 2013). This reduces GHG emissions during manure storage and fuel combustion if the CH_4 is utilised for heat and/or electricity generation. Additional benefits of anaerobic digestion include reduced water pollution, remainder can be used as fertiliser (with high mineral N content) and sharply reduced odours (Bates, 2001). The biogas has a composition of about 65% CH_4 and 35% CO_2 and is generated by bacterial fermentation of the organic content in manure/slurry which leads to conversion of 40% to 60% of the organic matter to biogas (Bates, 2001).

The efficiency of anaerobic digestion depends on several factors such as temperature, pH, carbon-to-nitrogen ratios, water-to-solid ratios, nutrient composition, particle size, retention time and quality of manure (Weiske et al., 2006). Sometimes for increasing biogas production, co-substrates as potatoes are added to the anaerobic digestion process. Although GHG emissions from manure storage can be substantially reduced, there is a risk of GHG release: i) during storage prior to the process of anaerobic digestion, ii) by CH_4 leakage from the anaerobic digester and iii) by emissions from digestate and residuals after the anaerobic digestion process. It is possible to distinguish between anaerobic digesters that are located in farms directly using manure/slurry produced at the farm or being centrally located in between surrounding farms and being supplied with manure/slurry from these. Latter usually have a higher capacity compared to on-farm plants.

2.3.3.2 Alternative mitigation options

The housing structure for the animals determines the possibilities for manure management in terms of storage, processing and utilisation of the manure. For instance, manure GHG emissions from swine being held on deep litter are 20% higher compared to holding at slatted floors (Montes et al., 2013). Housing systems should also allow for daily manure removal as these show generally lower manure GHG emissions. There have been several manure storage cover proposed including natural crust, wood chips, oil layers, straw and sealed plastic covers. The effectiveness of the cover depend on several factors e.g. permeability, thickness, degradability, porosity and management (Montes et al., 2013). Semi-permebale covers are a simple technology to reduce CH_4 and NH_3 emission from manure storage but can increase N_2O emissions. The mitigation potential is not clearly understood and science shows varying results on GHG reduction potentials. Manure acidification could also deliver CH_4 and NH_3 reduction potentials but it is not well assessed in terms of N_2O emissions e.g. when the manure is applied to the field.

2.4 Conclusion

The previous sections described the GHG emission sources from livestock production, their quantification and several mitigation options in livestock production. Enteric fermentation is the main GHG emission source from livestock production and could offer largest reduction potentials. Feed additives that change the diet for reducing enteric CH_4 emissions or decreasing emission intensity through productivity gains have been strongly discussed in recent scientific literature. Science is moving fast in this field and thus evidence is updated quickly. However, there are substantial knowledge gaps that need to be addressed in future research. For guiding mitigation policy design, the key question to be addressed is to what extent and at what costs the described mitigation options can contribute towards reducing GHG emissions.

Chapter 3 - MACCs and uncertainty

This method chapter describes MACCs and applied uncertainty assessment tools in detail. It includes the rationale for using MACCs as a policy decision tool, an elaboration of the engineering (ENG) MACC approach, description of developing an ENG MACC and the nature of uncertainty assessment for MACCs. Uncertainty and other methodological tools are elaborated in subsequent chapters.

3.1 Understanding the economic abatement potential

The technical abatement potential corresponds to the maximum abatement potential that is only limited by biophysical settings inherent to the system of interests. For instance, it describes the mitigation potential by applying mitigation options to all agricultural land or animals that are suitable and available for implementation. Assessing the technical abatement potential can be a preliminary step for identifying mitigation options, but is not sufficient for policy makers who seek to implement GHG abatement at lowest cost for society. For policy guidance, information on the CE of GHG abatement from different mitigation options is required for identifying the economic abatement potential corresponding to the optimal level of abatement. The optimal level of abatement is defined as the intercept of the marginal social benefit (MSB) and marginal social cost (MSC) of GHG emission abatement (Figure 3.1, Pearce and Turner, 1990). Figure 3.1 shows that investments into pollution control cause benefits to the society as impacts of pollution is reduced but also imply costs for the society for implementing mitigation options. The benefits for the society are greater than the cost up until the optimal level of pollution abatement. Pollution reduction beyond this point results in costs that outweigh the benefits to society and thus this money spent for mitigation could be better invested elsewhere. In other words, the MSC at the optimal level of abatement defines a threshold e.g. a carbon price that allows by not exceeding, economic efficient abatement. As shown in Figure 3.1, the technical abatement potential (full abatement potential) is considerably higher as compared to the economic potential (Schneider and McCarl, 2006; Smith, 2012).

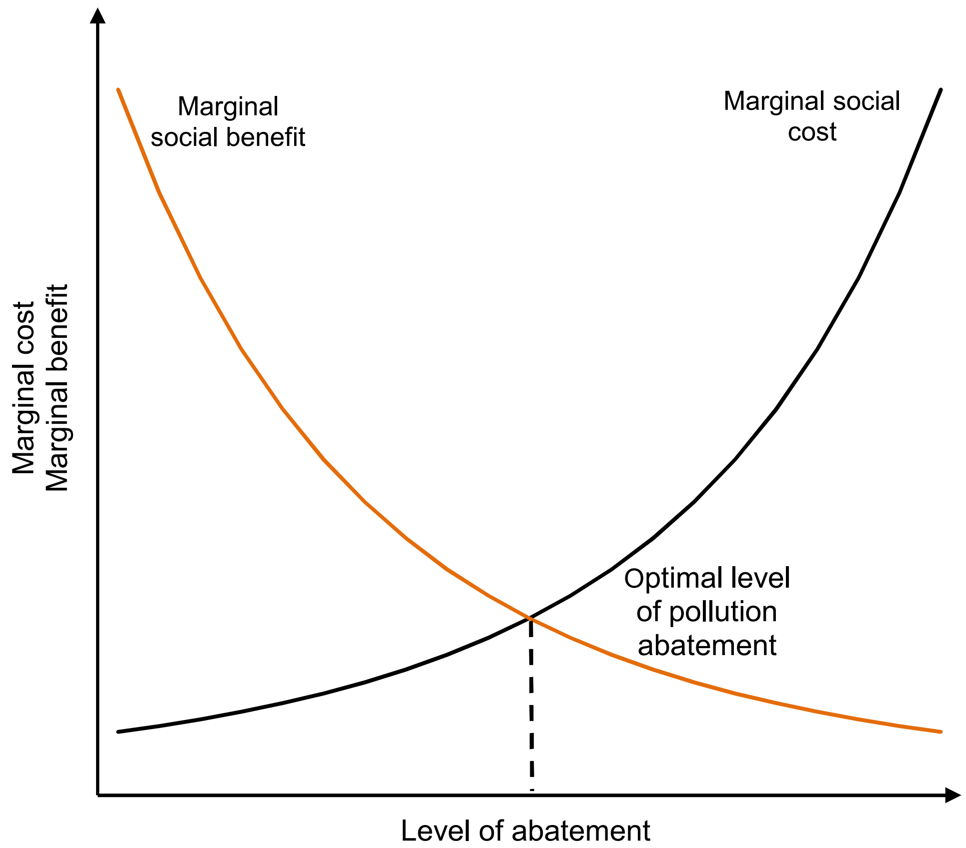

Figure 3.1: Optimal level of pollution abatement.

3.2 Marginal abatement cost curves

MACCs enable to identify the lowest costs of reducing GHG emissions by one additional unit usually in a future point of time. In other words, MACCs help to assess the optimal level of abatement, in which mitigation options are implemented to an extent that increases the marginal costs to only as much as desired (Klepper and Peterson, 2006). This clear definition of the mitigation costs can in turn convince decision makers to take action on climate change mitigation within the desired sector. The strength of MACCs is clearly their graphical design that presents the abatement potential in relation to costs in an easily understandable way. MACCs are applicable from farm level to global scale depending on the data that is available to construct them.

The use of MACCs dates back to the 1970s when they were developed as a response to the oil crisis. These so-called saving curves or conversion supply curves delivered information on options for increasing energy efficiency and thereby reducing dependency on oil (Wächter, 2013). An early example is the study of Meier (1982) that identified cost of reducing electricity consumption (in \$/kilowatt-hour). Since then MACCs were utilised for several purposes e.g. cost of abatement potential of air pollutants (\$/kiloton), cost of waste reduction (\$/kilogram), water availability (\$/meter^3) and cost of GHG emission reduction (\$/ton CO_2e; Kesicki and Strachan, 2011). The study of Jackson and Roberts (1989) might be the earliest example of using MACCs in a C reduction context and de Jager and Blok (1996) is one of the earliest studies that assessed cost for CH_4 reduction in the agricultural sector. However, the popularity of MACCs in the context of climate change mitigation has significantly increased since the reports of McKinsey & Company that published cross-sectoral MACCs for the USA (Creyts et al., 2007), globally (Figure 3.2; Nauclér and Enkvist, 2009) and for other countries, including Brazil, China, Germany, India, Russia and the United Kingdom amongst others[2]. MACCs have been promoted by various non-governmental organisations e.g. the World Bank that developed the MACtool which was utilised by Brazil, South Africa and Ukraine amongst others (Levihn et al., 2014). Today MACCs for climate change mitigation deliver a broad range of information including i) economic abatement potential of regions, countries, continents or global, ii) cost-effectiveness of mitigation options that are sector, regional or technology specific and iii) assessing the price of emission allowances from various sectors for e.g. Kyoto-based carbon trading (Levihn et al., 2014; Taylor, 2012).

[2] A complete list of countries for McKinsey developed MACCs is available at: http://www.mckinsey.com/client_service/sustainability/latest_thinking/greenhouse_gas_abatement_cost_curves

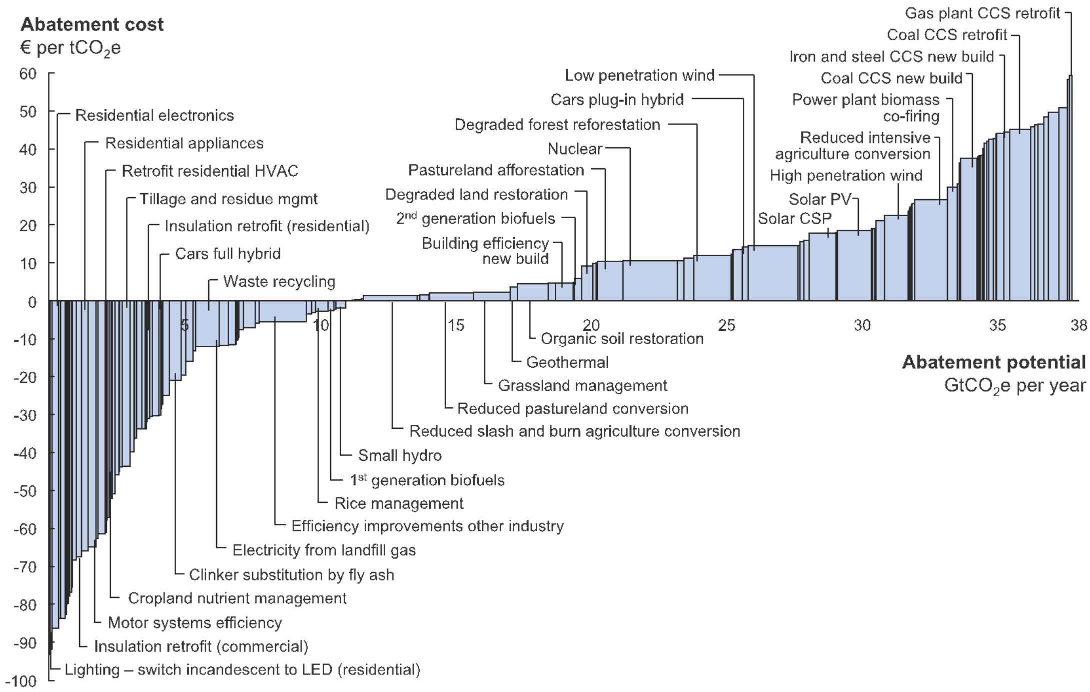

Figure 3.2: Global MACC from McKinsey in 2030.
Source: Nauclér and Enkvist (2009)

A MACC consists of two axes where the horizontal axis commonly represents the abatement potential e.g. in Mt CO_2e and the vertical axis depicts the CE of GHG reduction which is simply the ratio of implementation cost and abatement potential e.g. in €/ton (t) CO_2e (Figure 3.2). Many variations in MACC design are possible including MACCs that show relative abatement levels to baseline GHG emissions and those focusing on country specific emission targets e.g. Kyoto targets (Klepper and Peterson, 2006). With implementation of more mitigation options, the abatement potential increases from left to right while the CE decreases (Figure 3.2). The area below the curve shows the total costs of GHG reduction.

The shape of the curve can be either continuous or step-wise. For the latter, each step represents the implementation of one mitigation option. The advantage of a step-wise illustration is that CE and abatement potential are identified for single mitigation options i.e. the height describes the CE and the width the abatement potential of that particular mitigation option (Figure 3.2 and Figure 3.3). This enables the user to identify those measures which should be considered for achieving emission reduction targets and allows an easy identification of carbon prices for each mitigation option. From policy maker's perspective, a MACC can also provide information on ideal policy instruments that are required to achieve GHG reduction potentials. For this, the curve can be separated in different areas that indicate ideal policy instruments depending on the CE of abatement: i) for negative cost abatement, command and control political instruments should be used, ii) for cost-efficient measures (below the carbon price threshold), market based policies, i.e. taxes or carbon permits should be considered, and iii) high cost abatement potential requires further research and development or investment in e.g. infrastructure to make this abatement potential economically viable (Figure 3.3; Halkos, 2014).

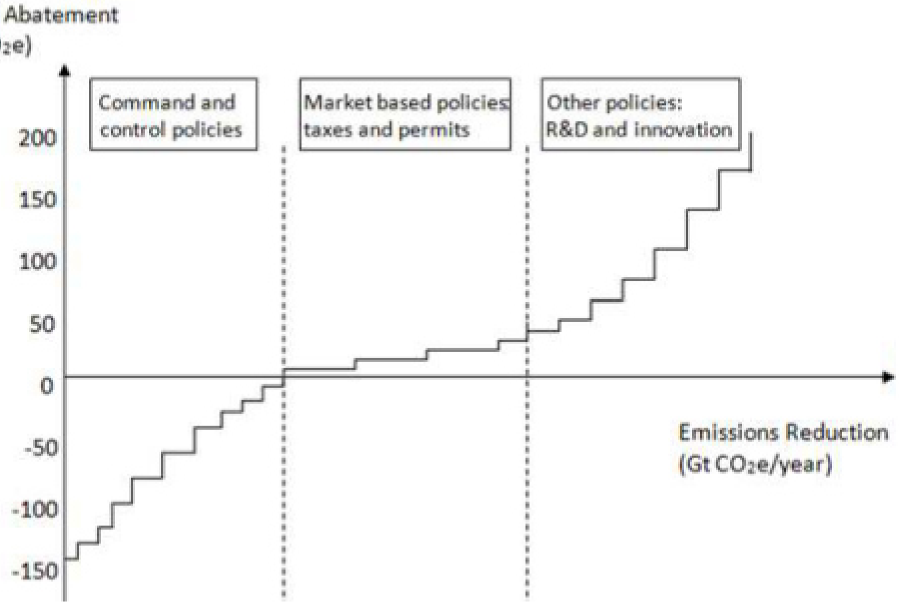

Figure 3.3: A schematic step-wise MACC separated by ideal political instruments to enforce mitigation potential.
Source: Altered from Halkos (2014)

3.3 Different MACC approaches

MACCs can describe either the full-potential or achievable-potential. Full-potential MACCs describe a situation in which a mitigation option is adopted at its full technical adoption potential (Wächter, 2013). This approach can be applied for instance for assessing the CE of total replacement of conventional vehicles by electronic vehicles with latter reducing emissions by roughly 4/5 per kilometre (Vogt-Schilb et al., 2014). For this approach, a full replacement is desirable as this technology involves massive investments costs for infrastructure and marketing. However, full-potential MACCs should usually be a preliminary step for policy design as these do not allow a comparison of other technologies targeting same abatement opportunities, and often it is not desired to switch entirely from one technology to another as regional implementation barriers vary considerably. Achievable-potential MACCs are the most common type. These MACCs describe a situation in which complete adoption of new technologies is not possible due to implementation barriers and other competing technologies (Vogt-Schilb et al., 2014). In the case of electronic vehicles, full adoption would be possible only in the long term, but not in shorter assessment periods. Further, implementation barriers are an important factor for the MACC simulation, and this is a controversial issue as data on such barriers are often not available. However, the consideration of an adoption rate for technology implementation is a major advantage of achievable-potential MACCs as this poses important information for decision makers and can also help to schedule mitigation policies correctly (Vogt-Schilb et al., 2014).

In the following part of this study, the focus is entirely on achievable-potential MACCs. However, there are many other typologies of MACCs available which can be differentiated in terms of their focus, research field, purpose and methodology. In this dissertation, MACC approaches are segregated based on different methodologies, including ENG, model-based and hybrid MACC approaches (Vermont and de Cara, 2010). Over the last two decades, ENG MACCs gained popularity and were developed for various agricultural systems in different countries (Bates, 2001; Graus et al., 2004; Moran et al., 2011; Pelletier and Tyedmers, 2010; Schulte and Donnellan, 2012; Smith et al., 2008; Wang et al., 2014). Model-based approaches are purely based on model simulations that define the system of interest by technical, political, behavioural and economic constraints based on standard economy theory. These approaches can be further segregated into supply side models (SSM; e.g. Breen, 2008; De Cara and Jayet, 2011; Durandeau et al., 2010; Hediger, 2006; Lengers, 2012; Smith and Upadhyay, 2005) and equilibrium models i.e. computable general equilibrium models (CGE; e.g. Golub et al., 2009; Hertel et al., 2009) and partial equilibrium models (PEM; e.g. Havlik et al., 2014; Key and Tallard, 2011; McCarl and Schneider, 2001; Reisinger et al., 2013; Schneider and McCarl, 2006; Schneider et al., 2007). Hybrid MACCS are a combination of characteristics from ENG bottom-up and macroeconomic top-down approaches (Jacobsen, 1998). In the agricultural context, MACCs derived by the Greenhouse Gas and Air Pollution Interaction and Synergies (GAINS) model can be classified as a hybrid approach as they combine profit maximising assumptions with measure-specific approach similar to ENG MACCs (e.g. Amann et al., 2008; Höglund-Isaksson, 2012).

While the ENG MACC approach is elaborated in the following chapter, a detailed description of the other approaches can be found in Appendix 1. Each MACC approach has its specific characteristics that are likely to impact the resulting abatement potential and CE. In comparisson, model-based approaches allow for a easier incorporation of model uncertainties and are capable to integrate interaction between the mitigation options and market feedbacks; thus can prevent from double counting of GHG emission reduction. ENG MACCs allow consideration of more mitigation options, measure specific estimates, assessment at higher detail level and are equipped to assesses heterogenity of a sector. This is particularly important for the agricultural sector as it is characterised by a high heterogeneity in terms of bio-physical and economical settings and innovative mitigation options should be identified. Further, advantages and disadvantages of the different approaches are discussed in the specific chapters. Results of model-based approaches are generally not comparable with ENG approaches (Vermont and De Cara, 2010). However, no approach is perfect and different approaches are required to accurately assess the cost of climate change mitigation (Schneider and McCarl, 2006). To reconcile the information generated by the different approaches, ENG MACCs could incorporate interaction and market feedbacks by implementation of specific mitigation options as estimated by model-based approaches. For this, current model-based solutions require to incorporate a larger set of mitigation options in their simulation. Further, hybrid models could combine the strengths of these two approaches but further research is required to improve such models particularly for the agricultural sector.

3.3.1 Engineering MACCs

ENG MACCs are bottom-up approaches that can be described as an assessment of individual measures. Bottom-up assessment is characterised by a preliminary focus on a smart part of the system of interest e.g. farm-level with subsequent up-scaling to the entire system. Usually, bottom-up approaches do not assume that the system is in an optimal state and hence system-wide interactions are not considered. There are various synonyms for the ENG MACC approach across literature including expert-based MACCs (Kesicki, 2010) or measure-explicit MACCs (Vogt-Schilb et al., 2014). Methodologies for ENG MACCs usually range from simply expert consultation to Excel spreadsheet exercises. Contrary to model-based approaches, single mitigation options are manually implemented into the system of interest to subsequently understand the impact on the system, based on a detailed technological and scientific understanding of the system and measures' impact. Such impacts include changes in activity levels, capital investments, fuel and electricity consumption, human labour input, operation and maintenance of a mitigation option (Halkos, 2014). The CE for each mitigation option is then calculated as an emission price that describes at which cost the particular option becomes profitable and this price remains constant in course of the full abatement of that measure (Vermont and De Cara, 2010). For the ENG MACC approach, it is of particular importance to have robust information on projections of the system under investigation, impact of measures on the system, including structural changes in the system, interaction between the mitigation options and adoption rate of mitigation options. This is due to the fact that market feedbacks through interaction, supply and demand change of the system components are not simulated. Therefore, ENG MACCs usually do not assume mitigation options that entirely change the system and rather focus on measures with low or medium impact i.e. no significant changes in crop area, crop yield, animal productivity, animal numbers, production inputs or crop allocation shifts (Vermont and De Cara, 2010). The graphical design ranks the mitigation options regarding their CE of GHG abatement following the assumption that most cost-effective measures should be prioritised by the decision makers. This leads to the typical step-wise MACC design (Figure 3.4). The abatement potential at a certain carbon price is then simply the cumulative abatement of all measures below that price threshold (Figure 3.4).

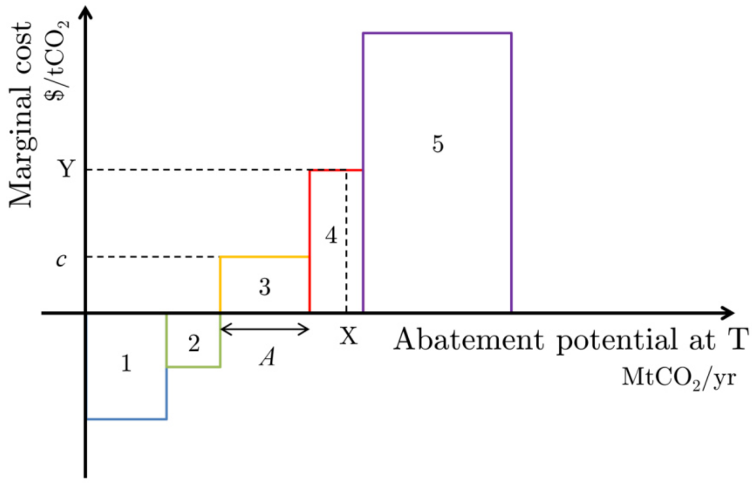

Figure 3.4: A step-wise MACC as an example for ENG MACCs. Each bar represents one mitigation option. The width of each bar shows the abatement potential and the height the CE of measure at a future point in time. The line starting from "Y" shows a budget constraint to a hypothetical mitigation target "X" i.e. the funding available to achieve a certain level of GHG reduction. Additional abatement potentials of mitigation options that show a lower CE then "Y" are currently too costly for this reduction target. The line "X" indicates also that only part of the total abatement potential of measure "4" is required to adhere to this mitigation target. Source: Vogt-Schilb et al. (2014)

In some cases, the ranking of mitigation options can be misleading. A prioritisation of mitigation options only by CE of mitigation options has the disadvantage that the absolute amount of potential GHG abatement is neglected (Wächter, 2013). For instance in Figure 3.4, the total abatement potential of measure 3 is lower as compared to the more costly mitigation option 5. However, the total abatement potential is an important variable in the climate change mitigation agenda, particularly in the light of ambitious GHG reduction targets as stated by the EU and China. This issue becomes more urgent if distinguishing between short- and long-term mitigation targets as current high-cost measures may be more attractive in the long run. Hence, it could be beneficial in starting to implement these prior to more cost-efficient options for preventing a 'lock-in' situation by considering only cost-efficient abatement (Vogt-Schilb and Hallegatte, 2014). Such a 'lock-in' situation may even hinder the introduction of currently not cost-efficient mitigation options and can increase their implementation costs (del Río González, 2008). Additionally, the currently costly mitigation options can lead to increasing technological and mechanisation levels in the agricultural sector and are therefore beneficial for increasing competitiveness in a future agricultural sector. The decision maker should therefore prioritise mitigation options with focus on the actual conditions of the system, long-term climate change mitigation and other political targets (Vogt-Schilb and Hallegatte, 2014). Generally for all MACC approaches, they only represent the CE and abatement potential for a future point in time, and the typical graphical design does not present the GHG emissions development prior to this. As proposed by Vogt-Schilb et al. (2014), ENG MACCs could be designed in a way that they show the adoption rate and corresponding GHG reduction potentials of different mitigation options. For this, the original MACC is flipped and aligned next to a wedge curve of the GHG reduction scenario induced by each mitigation option (Figure 3.5). This is valuable information for the decision maker since it discloses both the abatement and adoption rates for each mitigation option and hence allows for a more precisely timed planning of mitigation action. This is particularly important with regards to increasingly stringent mitigation targets. Implementation of mitigation options should be chosen in order to ensure meeting short-term mitigation targets without missing to invest in mitigation options for achieving long-term targets. Thus, the decision maker can prioritise the mitigation options without being solely dependent on a ranking by measures' CE (Figure 3.5). However, this graphical design does not describe how the costs will evolve in future. For instance, if investment costs are required for reinvesting into agricultural machinery or other indirect costs in case of path dependency. However, this poses a critique for the MACC approach in general (Kesicki and Ekins, 2012).

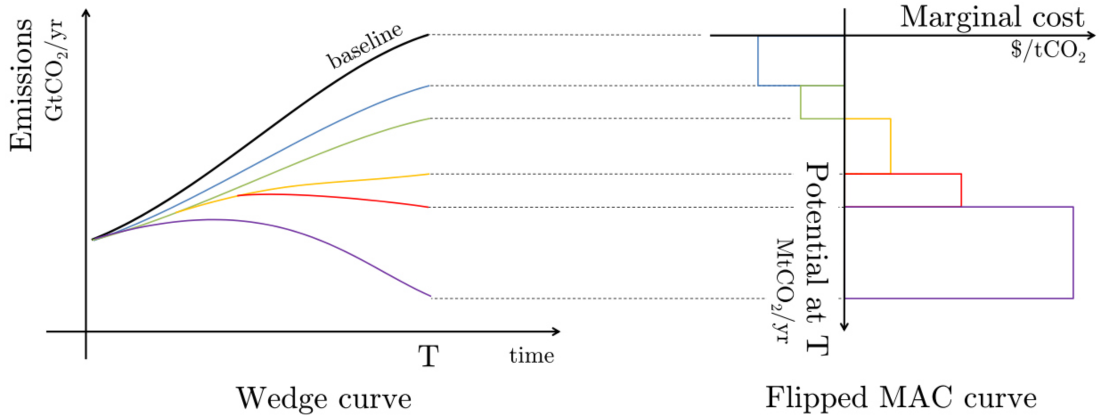

Figure 3.5: A combination of a 'flipped' MACC next to the corresponding wedge curve that describes the GHG emission reduction scenarios induced by the mitigation options.
Source: Vogt-Schilb et al. (2014)

Contrary to model-based MACCs, ENG MACCs allow for identification of negative cost abatement i.e. cost saving measures since they do not assume the economic sector to be in equilibrium and there have been critics raised that the economy is ever in equilibrium (see Appendix 1). In fact, a large share of the abatement potential reported by ENG MACCs is available at negative costs. At a first glance, it is noteworthy that such negative cost measures exists as one might think that negative cost potential or in other words 'getting something for nothing' would be already adopted in a competitive economic system (Taylor, 2012). However, this assumption is only valid in a real efficient market which does not necessarily reflect reality. It will therefore be explained as to why negative cost abatement occurs.

1) Farmers' decisions are not solely based on profit maximisation. A rational behaviour is influenced by cultural and social constraints (Moran et al., 2013). For instance, the social behaviour of humans shows that decision can be attributed to gaining positive attention from the social environment. This leads to adoption of prestigious management techniques that are visible to the outside i.e. conventional tillage. Mitigation options that focus on GHG emission reduction may not be visible and hence the urgency for implementation is lower (Moran et al., 2013).

2) There is no perfect information dissemination and farmers lack in knowledge of management techniques for reducing GHG emissions and skills of implementing these (Kesicki and Ekins, 2012).

3) A MACC is based on average values, but single farms show different economic and biophysical characteristics. Hence, the CE of a mitigation option varies strongly in farms within a heterogenic agricultural sector (Moran et al., 2013).

4) Farmers have financial constraints and funding new management techniques e.g. reduced tillage is a financial burden, especially when future income is uncertain (Smith and Olesen, 2010).

5) MACCs do not account for transaction- and monitoring costs for knowledge dissemination and large-scale implementation. However, these costs are significant and would lead to a higher CE and may delete cost saving opportunities (Moran et al., 2013).

6) Negative cost mitigation options may not be a sole technological choice. There could be the situation that despite a farm identifying cost savings, it cannot be implemented due to limitation posed by the market structure or confined market power of that particular farm (Bréchet and Jouvet, 2009).

7) Uncertainty of model design and input data may diminish the accuracy of ENG MACC results, thereby leading to negative cost measures.

8) The discount rate has a strong impact on the CE, particularly for MACC curves over a long time period (Kesicki and Ekins, 2012). Therefore, choice of discount rate can largely decrease the magnitude of abatement potential at negative cost. For instance, projects with high initial investment costs and low operating costs show only long-term benefits and their economic feasibility will only be favoured by a low discount rate.

There is a problem with the ranking of negative cost mitigation options that does not occur with positive cost options since the CE is the relation of implementation cost and abatement potential of a particular mitigation option (Taylor, 2012). For positive cost measures, a high CE corresponds to low implementation costs and/or high abatement potential, both of which are desired objectives. For negative cost measures, a high negative CE can be based on high cost saving potential and/or a low abatement potential. The latter is not a desirable objective and thus leads to a ranking of negative cost measures that does not represent the most economic viable mitigation options (Taylor, 2012). However, this is rarely the case, and the MACC user should be aware of this.

3.3.1.1 Advantages and disadvantages of engineering MAC Cs

An advantage of ENG MACCs is that they are easily understood and deliver measure specific information on abatement potential and associated costs. Considering single mitigation technologies allows specific actions in the system of interest e.g. a technology specific tax. Another advantage is the high technological and economic detail inherent to the ENG MACC approach. Therefore, this approach is well equipped for assessing heterogeneities corresponding to the structure of the system. This is particularly important for the agricultural sector as there is a high disparity between farms in terms of biophysial settings, size, structure and production output which subsequently lead to strongly varying levels of abatement potentials, applicability and implementation costs of mitigation options. Such detail level can further prevent technologically implausible results as observed with some model-based MACCs (Kesicki and Strachan, 2011). A detailed understanding of the sector usually leads to selection of a more justified set of mitigation options as compared to model-based approaches that comply well with the specifications of the farming conditions and are also more likely to be considered by farmers for implementing in their farms. Compared to model-based approaches, ENG MACCs are usually better equipped to deal with associated fixed costs resulting from changes in agricultural production by implementation of new agricultural practices and/or investment in these. This is due to the fact that high levels of detail allow an accurate assessment (Vermont and De Cara, 2010).

Although the measure-specific graphical design of ENG MACCs serves the purpose of being easily understood, the assumption that the CE of a certain mitigation option is constant for its whole abatement potential is not accurate in terms of structural, spatial and temporal heterogeneity of the agricultural sector. This graphical illustration can be misleading for the decision maker. ENG MACCs lack in assessment of market feedbacks that are caused by changes in market structure, mainly as this approach only focuses on a particular part of the economy and is restrained in providing a capable model to assess market feedbacks (Halkos, 2014). Market responses can be induced by mitigation option implementation which e.g. lead to higher milk production if the mitigation option affects milk yield positively or by policy decisions that influence the system of interest e.g. the abolishment of the milk quota system which is likely to change the structure of the European dairy sector. Further, abatement potentials derived by ENG MACCs can be subject to double counting or non-consideration of leakage as interaction within the system or influences of external systems are not accounted for. For instance, implementation of a mitigation option that focuses on GHG reduction from one source may increase GHG emissions from another source, which can be in- or outside the system of interest. Neglecting market feedbacks and interactions between different economies can be an obstacle for deriving a consistent baseline scenario and subsequent mitigation scenario, and this has implications for the robustness of the MAC estimates (see chapter 3.4). This may have consequences for designing effective mitigation policies if strong economical changes take place after measure implementation. Policies can for instance focus on mitigation options that are favourable at national level but increase total GHG emissions or mitigation costs globally. This would have particularly adverse impacts for global climate change mitigation. Therefore, the MACC developer needs to select mitigation options carefully and make the user aware of potential externalities. As discussed earlier ENG MACCs may be substituted by data from model-based approaches to account for market feedbacks correctly. By implementing mitigation options individually into the simulated system, assessing the interaction between the mitigation options can be obtained only manually. This can be subject to uncertainties as it is a major challenge to find accurate interaction factors that consider implementation of all proposed mitigation options simultaneously. In fact, not all ENG MACCs consider interaction between the mitigation options, which is one major criticism towards ENG MACCs (Levihn et al., 2014). Not considering interaction or wrong assessment can further lead to double counting of abatement potentials if some mitigation options target the same GHG reduction possibilities. Double counting of GHG reduction in a MACC must be avoided as this can misinform policy makers. Decision based on overestimated abatement potentials can lead to policies that focus on wrong mitigation technologies and thus may fail to achieve GHG reduction targets or create a 'lock-in' situation. Policy makers should therefore only use MACCs that do not include potential double counting. ENG MACCs also tend to neglect implementation barriers. Although recent studies account for the fact that full adoption of mitigation options is not feasible, the implementation of such barriers is exogenous and data on implementation barriers are not always available and have therefore to be obtained by expert judgement. Finally, the development of ENG MACCs is dependent on various information sources and includes a large number of variables such as measures' abatement potential, implementation costs of mitigation options, technology diffusion rate and discount rate. This makes an uncertainty assessment especially important to this MACC approach, but this is rarely done for ENG MACCs focusing on the agricultural sector.

3.3.2 Developing engineering MACCs

In this research study, two ENG MACCs will be developed i.e. for the EU-15 and China. There is not a single specific MACC methodology defined, and the methodological steps proposed here can vary from other ENG MACC approaches. Since both MACCs vary considerably in terms of spatial, temporal and agricultural factors, data and some variation to the general methodologies are unique to each MACC and are therefore elaborated in the specific chapter. The development of the two MACCs in this study can be categorised into the following six stages that are described below (Figure 3.6).

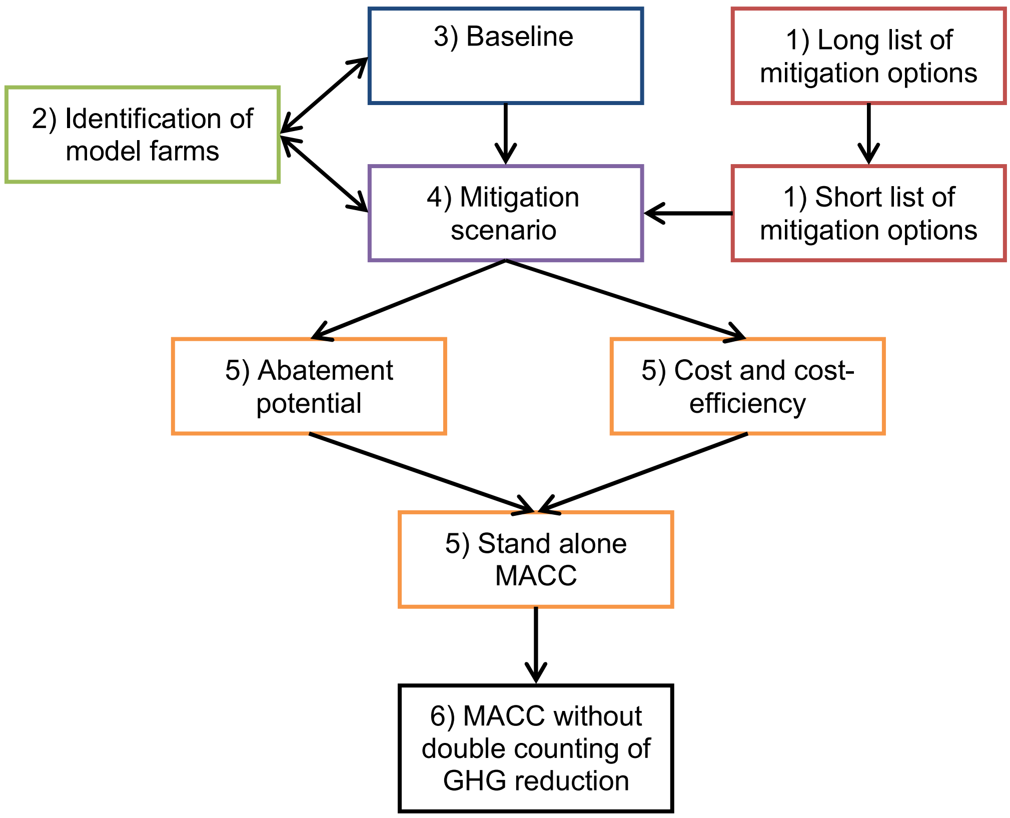

Figure 3.6: Schema of developing an engineering MACC.

3.3.2.1 Identification of mitigation options

Based on an extensive literature review, a long-list of mitigation options were compiled that could be potentially implemented into the agricultural system under investigation. For both MACCs, no consumer demand mitigation options were considered, although cutting meat consumption showed high abatement potentials (Westhoek et al., 2014). The initial list was shortened under consideration of following criteria: i) availability of reliable data, ii) high possible abatement potential, iii) technological feasibility, iv) high adoption rate, and v) ethical acceptability. These information were obtained from scientific literature and expert opinion. However, the MACCs developed in this research emphasise on mitigation through feed supplementation as these target the largest single GHG emission source from livestock production i.e. enteric fermentation and are not well assessed in terms of cost-efficiency of abatement in the scientific literature. As discussed earlier, selection of mitigation options depends strongly on availability of reliable data related to activity levels, mitigation potentials, baseline projections and economic data. In case of availability of more data MACCs can be updated. To prepare for the following steps, each mitigation option was described according to their function and impact on the agricultural system.

3.3.2.2 Identification of model farms

Model farms were identified that were representative for the desired agricultural system. These hypothetical farms were based on statistical averages. Model farms were separated into several farm types that were either separated by farm specialisation e.g. beef, milk, sheep or country's specification. The higher the variety of model farms, the better the assessment of the sector's heterogeneity in terms of social-economic and biophysical settings. For each of these model farms, the current activities including land use, livestock, input and output factors, management practices, biophysical settings and subsidies were assessed. To simplify a subsequent up-scaling to the total agricultural sector, these variables were based on statistical averages. The detail level of the farm descriptions was crucial as it determined the accuracy of measure's impact assessment on the farms. Further, an economic profile was developed for each of the model farms that described the gross margin in the benchmark year.

3.3.2.3 Baseline generation

For both the MACCs described in this study, the target year was 2020. Hence, the baseline scenarios were defined from the benchmark year to 2020. Several mid-term projections were available for the agricultural sector e.g. from i) the Organisation for Economic Co-operation and Development (OECD) and Food and Agriculture Organisation (FAO; OECD-FAO, 2013), ii) the Food and Agricultural Policy Research Institute (FAPRI, 2011), iii) the Directorate-General for Agriculture and Rural Development (DG-AGRI), iv) the Common Agricultural Policy Rationalised Impact (CAPRI) model and v) China's Agricultural Policy Simulation Model (CAPSiM; Huang and Li, 2003). These projections included basic agricultural variables such as activity levels, production levels and price development. To avoid inconsistency issues, only one projection was consulted for the baseline generation. It was advantageous to use projections from such sophisticated institutions as these consider developments at a global level, which in turn ensured that baseline projections accounted for influences from other economies. These projections did not incorporate all desired variables e.g. technological development, production input, management practices and climate conditions amongst others. Therefore, these knowledge gaps were substituted by other projections or expert judgement. Baseline GHG emissions were assessed by using IPCC methodologies combined with region specific EFs as cited in scientific literature for i.e. land use in ha, animal species or fuel consumption with sometimes considering temporal variability. The following GHG emissions were considered i.e. direct and indirect N_2O emissions from soils, CO_2 emissions from C decomposition in soils, CH_4 and N_2O emissions related to manure management and CH_4 emissions from enteric fermentation. To convert different GHG emissions to CO_2e, the common global warming potential (GWP) for CH_4=25 and N_2O=298 on a 100-year time horizon were applied based on the IPCC report (2006). According to the baseline scenario, the previously defined economic setting of each model farm changed and with it, the gross margin of the model farms in 2020. Such changes included technological development, labour input, fuel input, machinery use, production output, land use, livestock on farm, livestock feeding, fertiliser input and crop and animal productivity amongst others.

3.3.2.4 Generation of mitigation scenarios

Each mitigation option was manually included in the baseline scenario to generate several independent mitigation scenarios. For both MACC exercises, the abatement rates of mitigation measures were based on scientific literature, including meta-analyses of other studies i.e. MitiGate[3] (Veneman et al., 2015), Lewis et al. (2013)[4], Nayak et al. (2015) and sometimes expert judgement which reported reduction potentials that were specific to Europe or China. The abatement rate can be defined as the GHG reduction potential per ha of land use (in tCO_2e ha^{-1} $year^{-1}$), animal (in CO_2e $head^{-1}$ $year^{-1}$) or C sink potential for increased C sequestration (in C ha^{-1} $year^{-1}$) by replacing a conventional management technique operating during the baseline. Initially, the total stand alone technical abatement potential was estimated, which was only constrained by biophysical limitations and was additional to baseline management techniques. After this, measure specific adoption rates until 2020 were introduced based on a mix of literature review and expert judgement. The adoption rate described the behaviour of the farmers and their willingness to adopt new management techniques. Therefore, the adoption rate defines increasing implementation levels over the projection horizon and most importantly the implementation levels at the final year of MAC assessment. GHG reduction potentials were only applied to this constrained agricultural land or animals. For the abatement costs, the mitigation options were individually introduced to the previously defined model farms, but only to those that applied the mitigation option. Based on literature and expert judgement, since data on costs was limited, all impacts of the measure on the farm were identified and applied to the baseline operation and economic profile of the model farms. Depending on the specific mitigation option, measures' implementation required investment costs, changed production inputs (including labour, machinery, fuel, fertiliser, seeds, feed, pesticides, electricity and fuels) and increased or decreased production output if yields are affected. Based on these factors, the gross margin of a farm with implemented mitigation option was estimated.

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[3] The MitiGate database is based on a meta-analysis of literature reporting for ruminant CH_4 mitigation through feed additives and dietary manipulation. It currently consists of 320 papers covering in vivo mitigation data for different world regions, animal types and production systems. Analysis was restricted to those studies which expressed methane emissions per unit of intake.

[4] The meta-analysis by Lewis et al. (2013) included 302 individual studies that focused on 246 different feed additives. This study gathered information on feed additives and their potential for reducing the environmental impact for livestock production (including GHG reduction potentials).

3.3.2.5 Stand-alone abatement potential and cost-effectiveness

After the generation of the baseline and mitigation scenarios, both scenarios were compared with each other. The difference between baseline and mitigation scenario was the abatement potential (equation 1):

$$Abatement\ potential_{measure} = GHG\ emission_{baseline} - GHG\ emission_{mitigation\ scenario} \tag{1}$$

The estimation of CE combined the emission scenarios with the discounted farmers' revenues during the baseline scenario and counterfactual mitigation scenarios. First, the future revenue after measure implementation was converted into the monetary value of the benchmark year of the MACC which was equal to the net present value (NPV; equation 2).

$$NPV = -C_0 + \sum_{i=1}^{T} \frac{C_i}{(1+r)^i} \tag{2}$$

With NPV being the net present value; C_0 described an initial investment that was in the MACC derivations equal to zero since all costs were divided by their lifespan to estimate annual costs; C_i was the annual cost that referred to the year 2020; and r is the social discount rate that varied considerably for the EU-15 and China.

Finally, the stand-alone CE for each mitigation option was simply estimated by dividing total costs by abatement potential for the particular mitigation option (equation 3).

$$CE_{measure} = {}^{discounted\ cost_{measure}}\!/_{abatement\ potential_{measure}} \tag{3}$$

For both MACC exercises, the ancillary costs and benefits were excluded. As discussed earlier, the first includes the cost for transaction, policy implementation, knowledge dissemination and externalities of the mitigation option, the latter included social benefits by replacement of the conventional management practice.

3.3.2.6 Simultaneous implementation of different measures

Potential double counting of GHG reduction was considered during simultaneous implementation of mitigation options. As discussed earlier, applying interaction factors is subject to large uncertainties and can lead to assessing reduction potentials incorrectly. Therefore, mitigation options were applied mutually exclusive for these that were potentially interacting i.e. adoption potential of one measure did not overlap with the adoption potential of another mitigation option. For instance, simultaneous application of different feed additives to the livestock is unlikely to cause cumulative effects on GHG reduction or yields (Hristov et al., 2013). A farmer would therefore not apply different feed agents to one animal for GHG reduction. This final step in the MACC derivation is expected to lead to a significant reduction of the abatement potential for some mitigation options, and this issue will be discussed in the following sections of this dissertation.

3.4 Critique of MACCs

This section elaborates critiques of the general MACC approach as literature on MACCs lacks in detailed critics of the usage of MACCs for climate change mitigation (Levihn et al., 2014). It is important to note that this list is not complete as it should rather highlight the controversy of using MACCs as a policy decision tool. The specific sections are not solely independent and may also interact in some regards.

Most MACCs approaches are similar in terms of focusing only on private costs of climate change mitigation without considering social cost of mitigation actions. This can be disadvantageous for decision makers as private costs differ from social costs, and the latter involves all externalities e.g. society's damage from pollution that a producer does not pay for**.** However, if ancillary costs pose a large burden to society, under-representation of externalities can be disadvantageous for a cost assessment, particularly in the climate change context. In regions that are negatively impacted by climate change and have only limited capacities for adaptation to climate change, it would be therefore useful that MACCs report costs of climate change (and other externalities) to the society. This would allow a better linkage between climate change mitigation and poverty alleviation. However, this is a demanding exercise due to large uncertainties of social damage cost in the climate change context. Failing to consider climate change impacts may be critical for MACCs describing the long term future but not for short term MACCs that have been developed in this research study. Regarding ancillary benefits and costs, the mitigation option 'fertiliser input reduction' is a good example as it not only reduces N_2O emissions, but also reduces N losses through leaching and NH_3 emissions and subsequently increases ground water quality. In fact, groundwater pollution is an acute problem in over fertilised croplands in many parts of the world. More generally, fertiliser reduction induces ancillary benefits for the society that are not accounted for in the common MAC assessment. In this context, Reis et al. (2005) raised awareness of linking GHG emission reduction with other external effects on environmental pollution. Since then various attempts to integrate ancillary impact in cost assessment of mitigation options have been undertaken. For instance, Wagner et al. (2012) showed the co-benefits of GHG mitigation on regional air quality while considering GHG emissions reduction simultaneously to reduction of sulphur, nitrogen oxides (NO_x) and particulate matter emissions in ANNEX I countries with subsequent improvements in human health and cost savings for air pollution control mechanisms. A MACC can be modified to account for social costs of abating multiple pollutants (GHG and non-GHG pollutants) if all pollutants can be defined in monetary terms to enable a direct comparison between them. Eory et al. (2013) simulated effects of integrating co-effects of mitigation options from the ENG MACC curve developed for the United Kingdom (Moran et al., 2008). The study stated that inclusion of non-GHG pollutants can change the CE of the mitigation measures depending on the damage cost chosen for these non-GHG pollutants. Meanwhile higher damage cost makes some mitigation options more cost-efficient and lower damage costs does not change measure's CE strongly since GHG emissions dominate total damage costs. This study further concluded that data availability on externalities for mitigation options is limited and further research is required to expand this knowledge base, also in terms of monetisation of non-GHG pollutants. Multiple pollutant MACCs are advantageous over the common approach in the regard that ancillary benefits of GHG emission abatement can be shown which could increase CE of some measures that were not cost-efficient before. This economically justifies more investment into mitigation activities. However, for assessing all ancillary effects, impacts on biodiversity, soil quality, human health, animal health, welfare and food security amongst others should be considered (Eory et al., 2013), but that poses a difficult and demanding task. It is important to note that the consideration of social costs from mitigation action could also reduce measure's CE. In this regard, Kesicki and Strachan (2011) stated that MACCs are weak tools for assessing indirect costs, non-financial costs, and also market failures and barriers. Indirect costs can relate to research and development for innovations, monitoring and evaluation of the measures' implementation process, policy implementation costs including costs for administration and/or knowledge dissemination. These hidden costs can be high for measure implementation. Kesicki and Ekins (2012) stated that transaction costs can range between 9% and 40% of total investment costs of a new technology in the energy sector which in turn is depending on the size of the project. Therefore, it is important to meticulously consider such costs as they are required for a successful technology introduction, particularly in markets in which farmers are not well informed. Failure in information dissemination could lead to non-adoption or adoption of non-optimal mitigation options where in the latter case, cost-efficient mitigation options could be missed, and more costs could arise if newly adopted management techniques have to be replaced by more efficient mitigation options (Kesicki and Strachan, 2011). For the agricultural sector, the market distorting influences of subsidies should be also considered as farmers may not adopt new technologies e.g. different crop types with a subsequent loss in subsidies. That is why farmers cannot be expected to act as rational agents in terms of economic efficiency. Hence, to increase incentives for farmers, restructuration of subsidies are required which can lead to additional cost. Further, implementation of new technologies is always subject to barriers, and diffusion of new technologies does not happen immediately, particularly in the agricultural sector where structural changes are gradually occurring. In this context, MACCs can be criticised as the adoption rate of new technologies may either be not accounted at all or only limited. The adoption rate is an important variable for measure implementation and hence the overall efficiency of that mitigation option (Wächter, 2013). For instance, slow adoption of new technologies increases implementation costs including higher marketing efforts, and the cumulative abatement potential can significantly decrease over time. Further, MACCs do not account for varying adoption rates as some farmers may adopt new technologies more quickly than others. Some MACCs also do not account for variation in abatement costs across the system, but farms vary in different regions in terms of income, job, resources access, education, cultural and financial settings (Casillas and Kammen, 2012). Assessing these heterogeneities would be particularly important for mitigation activities in the developing world as for most farmers' financial resources are limited. This can also apply for developed countries as different financial burdens to single farms require different mitigation actions. Additionally, spatial disparities could have an accumulated effect if ancillary costs and benefits are considered as these also largely differ amongst farms. Depending on the detail level of simulation, MACC approaches could more or less well consider heterogeneity in terms of abatement, mitigation costs and technology diffusion rate. As discussed earlier, ENG MACCs are better equipped to do so as compared to model based approaches as they generally inherent a high technological understanding of the system under investigation.

A discount rate is applied to compare cash flows in varying points of time. However, the choice of the correct discount rate is a critical issue which has been extensively discussed in the scientific literature since the 1960s (Feldstein, 1964). Generally, a high discount rate favours projects with investments in far future and financial gains in short term; contrary to a low discount rate that favours long-term projects which may have a large investment in early years and deliver financial gains only in the long-term. This is an important issue as costs of climate change to society are difficult to assess and involves high levels of uncertainty. It is further necessary to distinguish between a social and private discount rate. The social discount rate considers the perspective on the society i.e. an investment that may be beneficial for society, while latter focuses only on the private benefit (Kesicki and Strachan, 2011). The social discount rate is usually much lower compared to the private discount rate since private companies are not privileged to borrow money at cheap rates as the government and the risk involved in a project is more adverse for private enterprises and hence private investment requires a higher rate of return. A commonly applied social discount rate in the UK is 3.5% as recommended by the UK government, based on the extent to which future consumption values are less compared to the present. This is because wealth increases and is based on a hypothetical interest rate for governmental climate change mitigation projects (Kesicki and Strachan, 2011). Some MACC approaches apply discount rates that are sector or technology specific, and these can differ considerably from the social discount rate. However, most of the MACCs apply a social discount rate, although they include private costs of mitigation. Hence, individual farms are assumed to invest in mitigation options but in reality private investments apply a higher discount rate (Kesicki and Strachan, 2011). Application of the social discount rate makes sense to assess the social perspective i.e. the increase or decrease of welfare or impact on the economy, but this may not be in conformity to the behaviour of farmers in an agricultural market. As stated before, this has a significant impact on MACCs focussing on the long-term future.

MACCs can be criticised for their poor treatment of uncertainty (see definition of uncertainty in section 3.5). This is particularly true for ENG MACCs in the agricultural sector as model-based MACCs sometimes present uncertainty ranges of MAC estimations. Missing uncertainty assessments can be explained by the high heterogeneity of the agricultural sector and the challenge posed by the temporal and spatial variability in this sector. There are further reasons why science neglect uncertainty assessments and these are discussed in section 7.5.2. Since MACCs are a communication tool between scientists and policy makers, assessment of these inevitable uncertainties should be part of the MACC exercise to avoid miscommunication between climate experts and non-experts, and that could lead to erosion of trust and increases the probability that science fails to lead governmental decisions (Pidgeon and Fischhoff, 2011). Without reporting uncertainties, MACCs could be judged as a perfect snapshot of the future and mislead decision makers with potentially adverse implications for mitigation policy design. However, every CE analysis in future is subject to uncertainty as future projections cannot be validated by a real situation. In fact, MACCs are very sensitive to wrong assumptions and all variables including in- and output prices, activity levels, abatement potentials, technology development, adoption potentials, measure impact, baseline development and discount rate are subject to uncertainty. For uncertainty regarding future projections, the baseline scenario is of crucial importance for the MACC exercise. The baseline scenario is the basis of the mitigation scenarios, and the abatement potential and corresponding costs are derived by comparison of both these scenarios (Paltsev and Capros, 2013). Uncertainties within the baseline scenario would therefore propagate throughout the MACC exercise. However, to avoid extreme and uncertain assumptions, baseline scenarios usually follow a 'current policies' assumption with no major events in the economy being assumed, and this applies throughout the projection horizon. It therefore generally applies, that the longer the projection horizon, the higher the uncertainty that the 'current policy' assumption remains true. This is particularly important with regards to mitigation policies since increasingly stringent GHG reduction targets over time enforce stronger political action which affects the baseline scenario and subsequent abatement potential and costs of the mitigation options. 'Current policy' scenarios do not consider these potential political influences on the market (Paltsev and Capros, 2013). Also 'current policy' baseline scenarios have to deal with the issue that implementation level of current policies varies both currently and in future, and this has strong implications on the baseline scenario. Therefore, Paltsev and Capros (2013) assumed to use a 'no policy' baseline in which no mitigation policies are operating to estimate the cost of mitigation. This could pose problems by not estimating the real cost of mitigation and abatement potentials as policies are operating in real world. The prediction of future prices is also crucial for the baseline scenario and can strongly alter MAC estimations. For instance, fuel prices can influence abatement potentials and associated costs as increasing fuel prices favour the implementation of less energy intensive technologies and vice versa (Wächter, 2013). Due to the static nature of MACCs, they cannot be easily adjusted to changes in input variables (Kesicki and Strachan, 2011). This is a more adverse problem for ENG MACCS as compared to model-based approaches, as latter can be more easily adjusted. Input variable adjustment is important for price developments as their inherent variability might not be predictable, as experienced with the fuel price drop in recent times. Therefore, McKinsey updated their global MACC in 2010 to account for changes in fuel prices amongst others after stabilisation of the global economy following the global economic shock (Enkvist et al., 2010). Generally, variation of input variables should be considered for ENG MACCs, particularly those which cover large regions and subsequently large agricultural sectors as price elasticity (induced by mitigation action or unpredicted influences) can have major impacts on the MAC estimations.

There are several tools available to assess such uncertainties of MAC estimates, also particularly for uncertainties of future projections. It is important to evaluate uncertainty of MACCs, improve robustness of the simulation and increase awareness of decision makers to these issues (Kesicki and Ekins, 2012). Therefore, such uncertainty assessment of MACCs will be an important part of this research study.

3.5 Uncertainty in MACCs and assessment methodologies

The term uncertainty can be coined as the quality level of produced knowledge and defined as a situation with incomplete knowledge that is based on inexactness, unreliability or ignorance (Walker et al., 2003). Although the term uncertainty is widely used, understanding of uncertainty varies depending on the scientific discipline. In finance, uncertainty describes a situation in which more than a single outcome is possible and the probability of occurrence of this possibility is unknown. In decision making uncertainty relates to an unknown nature, unpredictability of consequences, magnitudes or events and probabilities that outcomes will not happen as predicted. In environmental management uncertainty “refers to a situation in which there is not an unique and complete understanding of the system to be managed” (Raadgever et al., 2011). Uncertainty can be roughly divided into two categories: i) irreducible uncertainty that is attributed to variability or randomness inherent to a system e.g. temporal or spatial variability without experimental errors and ii) reducible uncertainty or epistemic uncertainty arising due to lack of knowledge i.e. no variation within the system and with only experimental errors (Roy and Oberkampf, 2011). According to this definition, different levels of these attributes can be assigned to uncertainties i.e. ranging from purely irreducible over partly irreducible to purely reducible (Matott et al., 2009). Uncertainty analysis can identify and sometimes quantify the extent of uncertainty in a MACC model e.g. regarding abatement potential or CE. The characteristic of uncertainty here plays an important role as not all uncertainties are reducible after identification, but others can be reduced. Increasing the knowledge about uncertainty does not necessarily eradicate uncertainty, and may in fact increase understanding of uncertainty that leads to identification of previously unknown knowledge gaps.

Uncertainty assessment is of particular importance for model simulations that support decision making processes since models are rarely in perfect conformity with reality (Willows et al., 2003). Scientists and decision makers become increasingly aware of this fact as uncertainty has direct implications for the usefulness of model outputs (Allen et al., 2007; Clancy et al., 2010). Lack of uncertainty assessment can be a key reason of failure of model predictions and may lead to undesirable decisions. It is therefore a crucial exercise to precisely inform decision makers and increase robustness of the decision based on the model outcome (Lempert and Schlesinger, 2000). In this context, the purpose of uncertainty assessment is to reduce adverse effects of simulation uncertainty and allow for good decision making. Some of the most challenging issues in science and politics that are highly dependent on 'accurate' model simulations are climate change, economics of climate change, sustainable development and future energy supplies (Bastin et al., 2013). In this regard, uncertainty of MACCs should be assessed since the cost component justifies action or inaction for climate change mitigation. Uncertainties arise with regards to the natural variability and incomplete understanding of biophysical, economic, political and societal conditions. These factors strongly determine agricultural activities in terms of production levels, farm management, price developments, subsequent GHG emissions and abatement potentials and associated costs. For instance, the weather and soil conditions have a large impact on the crop type used in crop systems, but both these variables are subject to variability that is often irreducible. ENG MACCs that cover large regions have to deal with large spatial and temporal variability and hence heterogeneity inherent to agricultural systems. This can increase uncertainty if there is no adequate detail level of assessment. The interdisciplinary nature of ENG MACC models and hence the reliance on various information sources could further potentially accumulate uncertainties without awareness of the MACC developer. This is exacerbated as large variability in spatial and temporal scales requires a large amount of model variables and parameters, and combining these in a model simulation can easily be sources of uncertainty. This can be particularly adverse for MACC exercises as there is usually no reliable information on uncertainty for all input variables and hence this is a hindrance to accurately assess uncertainty. For historical data (e.g. price development), the upper and lower bound of uncertainty can be defined, but for data without availability of such data series (e.g. discount rate, adoption potential and GHG reduction potentials) defining uncertainty can be a demanding task. In case of MACC models that are not linear or deterministic, uncertainty of model inputs can bias the output in an odd and incomprehensible way, and thereby making identification of uncertainties even more crucial. MACCs usually allow information on uncertainties to be accommodated that can be quantified e.g. a range of possible CE's for individual mitigation options. However, quantifying all uncertainties that influence model output remains an ideal situation and can be rarely achieved as it requires a large input of resources in terms of working hours and computing capacities. It is thus helpful to identify different sources of uncertainties in the MACC design to focus the uncertainty assessment only on key uncertainty sources. In addition to this, MACCs should also report uncertainties that cannot be quantified in a qualitative way.

3.5.1 Sources of uncertainty

There are various definitions of uncertainty sources with discrepancies amongst them depending on the scientific disciplines and their purposes (Bastin et al., 2013; Refsgaard et al., 2007; Roy and Oberkampf, 2011; Trucano et al., 2006; Walker et al., 2003; Willows et al., 2003). Not all of these definitions focus on model-based decision support and only fewer are available for economics of climate change mitigation. Despite this inconsistency in terminology, this section gives a broad overview on categories of uncertainty sources relevant for MACCs and includes information from various studies. Not all categories are mutually exclusive and can overlap in some regards. In this section, the uncertainty sources are discussed from the modeller's perspective i.e. focus on the accumulation of uncertainties during the model simulation process as reflected in the model outcome and hence the robustness of the model output to support decisions rather than from the decision maker's perspective (Walker et al., 2003).

1) *Uncertainties about reality* - As stated earlier, variability inherent to the system e.g. climate variability, frequency of extreme events in terms of climatic, economic or social parameters and behaviour of individuals cannot be simulated with 100%

accuracy. Such approximation is a potential source of uncertainty (Willows et al., 2003).

2) *Knowledge uncertainty* – Knowledge uncertainty particularly describes incomplete knowledge about certain simulated and experimental data. First, source of uncertainty is due to incomparability with real data e.g. simulations of future development that can only be approximated (Willows et al., 2003). For experimental data, there is always uncertainty due to experimental error as experiments always influence the system of interest and hence bias the results. However, uncertainty in future projections is of particular importance for MACC models as they simulate the cost of GHG abatement at a future point in time.

3) *Model uncertainty* – Model uncertainty can be defined as model limitations that limit to precisely reflect reality. This can be due to incomplete data, inconsistent structure, irrationality and limited reflection of the system of interest (Willows et al., 2003). Since this is of high importance for MACC simulations, it is distinguished between following locations within a model uncertainty.

    a. *Model context* – The model context is set to describe the system of interests and can show uncertainties in terms of accurate reflection of the system. Context uncertainty includes uncertainty about environmental, political, social and technological conditions that define the problem under investigation (Walker et al., 2003).

    b. *Model structure* – Limitation due to model structure are inherent to each model simulation. Model structure uncertainties can arise from lack of understanding of the system of interest including the system's natural behaviour and interactions between the systems' elements (Walker et al., 2003). Technical components e.g. software bugs, typing and algorithm errors also come under this category and can lead to further uncertainties. The choice of model can have a large impact on the reliability of the model output; hence the model is crucial and should be chosen carefully (Willows et al., 2003). Good examples of this uncertainty source are the varying climate scenarios from different climate models, although some of these models are based on same data sources and fundamental physics.

c. *Model input* – This category includes all model inputs i.e. variables, parameters, mathematical functions that are fed into the model simulation. Uncertainty arises from a lack of knowledge and natural variability of model inputs. These uncertainties can be further increased by 'external driving forces' that influence the system of interest and are not controllable by the modeller (Walker et al., 2003). For ENG MACC models, important uncertainty sources are associated with future projections of e.g. prices, activity levels, spatial, adoption potentials of mitigation options and temporal and natural variability. Although parameters are mostly based on experimental results that should limit uncertainties to a certain extent, spatial, temporal and natural variability can be still a source of uncertainty, particularly for MACC models.

d. *Model output* – Uncertainties in the model output are the accumulated uncertainties from the above mentioned sources. If model output can be compared to real data it is called 'prediction error' (Matott et al., 2009). In case of MACC model output, these cannot be compared to real data and this requires a specific type of uncertainty assessment (as discussed below).

3.5.2 Tools for uncertainty assessment

Although there are various tools for uncertainty assessment, this section discusses only the tools SA and MC simulation as these tools will be used in chapter 4 and 6, respectively. For a brief description of other tools, refer to the Appendix 2. Deterministic approaches e.g. SA define a discrete value for input variables or parameters while probabilistic approaches e.g. MC simulation assume a range of probabilities for model inputs. Generally speaking, probabilistic approaches are more complicated but also more sophisticated, since they assess probabilities and determine the range of possible model outputs.

3.5.2.1 Scenario analysis

SA can assesses various future scenarios and aims to explore consequences of these scenarios to support decision making processes (Refsgaard et al., 2007). Scenarios can be defined in various ways such as: i) consistent and challenging descriptions of possible futures, ii) potential future situations to inform decision-making under conditions of uncertainty or iii) fundamentally different futures presented in a coherent way (Reilly and Willenbockel, 2010). The emphasis is clearly on multiple possible futures that describe different 'what-if' scenarios. SA is an important tool to extend limitations that arise by only considering one possible future in decision making processes. Taking into consideration multiple possible futures may increase awareness of prediction uncertainty, response capacities and robustness of policy decisions with regards to climate change mitigation efforts (Duinker and Greig, 2007). In terms of uncertainty assessments in ENG MACCs, SA can increase the possible model output space and thereby allows evaluating the ranking of mitigation options based on different baseline scenarios which is of crucial importance for assessing GHG abatement potentials and CE.

For model simulations, SA originated from the Manhattan Project in 1942 and was applied to decision-making based on simulation of atomic explosions (Reilly and Willenbockel, 2010). During the 1960s, studies considering future scenarios were also utilised for non-military purposes e.g. for understanding economic, political and technological future developments in developed economies in order to react accordingly (Berkhout et al., 2002). The book "The Year 2000" serves as one of the earliest examples (Kahn and Wiener, 1967). For SA application in the corporate sector, the company Shell exemplifies a leading role in the pioneer use of SA as strategic planning tool in the early 1970s as a range of possible futures were considered that could have implications for the company (Swart et al., 2004). SA increasingly gained popularity after the oil crises of the 1970s and particularly for strategic business planning. Currently for future planning, SA is applied in various fields including environmental assessment and community management. SA is of particular importance for climate change science as future climate is subject to large uncertainties due to its dependence on many factors. For instance, the IPCC reports include a range of possible future climate scenarios based on different models holding different climate sensitivities and GHG emission trajectories.

SA can apply uncountable different scenarios which can be quantitative or qualitative based on model simulations in case of former and on expert opinion in the latter case. However, it is beneficial for the decision maker that the “what-if” scenarios cover a certain range of possible futures that could be based on the following assumptions: worst case, best case, business as usual, no change, best guess or historical development (Willows et al., 2003). The number of scenarios applied should be around 3 to 5 as fewer numbers could lead to an under-representation of possible futures and more scenarios could create an overwhelming amount of possibilities for the decision maker (Amer et al., 2013). It is important to mention that each scenario should be treated equally. Several typologies for scenarios are available for explaining their respective purposes. A modified classification by Börjeson et al. (2006) is elaborated below:

1) Predictive scenarios aim to predict the future as accurately as possible. Such scenarios are used for strategic planning and decision making that relies on foreseeable future situation for increasing awareness of potential changes and associated opportunities. These scenarios usually assume no changes in policies during the projected horizon and often rely on historical data series. Sub-categories include forecast scenarios i.e. presenting the most likely future development and what-if scenarios i.e. showing the consequences of near future events over the projected horizon. Forecast scenarios are most suitable for short-term projections and are of particular importance for strategic business planning. As discussed earlier, most MACC baselines can be assigned to this category. The what-if scenarios show the impact of one or few variables that can change due to an external or internal event. Here the probability of that scenario has lower priority than presenting the impact of the changing variables on the system. A key disadvantage is that they do not account for novel or surprising changes or more drastically speaking, they often assume the future is a continuation of the past (Berkhout et al., 2002).

2) Exploratory scenarios help to explore the future in situations where variability is expected and hence strong uncertainty regarding the future development. These scenarios therefore help to explore consequences of alternative developments that can lead to a range of possible futures. They describe a situation where the actor has limited influence on the development of the system and thereby can be used to understand the capacity of the actor to adapt to a new situation. Being primarily based on qualitative data, these scenarios are hypothetical and fairly general (Reilly

and Willenbockel, 2010). A major disadvantage is that this scenario type could be unrealistic and hence not suitable for strategic planning.

3) Normative scenarios are based on storylines leading to a specific target that can be based on a positive or negative futuristic vision. Beginning from a normative starting point, the focus is on how to realise a certain objective in future. The differentiation is made on the basis of the system's structure and how the target can be achieved by either preserving or transforming the system's structure, provided that in the latter case the original system structure does not allow an efficient achievement of the target. These scenario types are usually the least predictive and can be compared to objective based planning that includes setting milestones and actions (Reilly and Willenbockel, 2010). The key disadvantage of these scenario types is that they overrate the ability of the actors to take action or that the future develops in this particular way (Berkhout et al., 2002).

3.5.2.2 Monte Carlo simulation

MC is a commonly applied statistical technique to evaluate the impact of model input uncertainty and error propagation during the model simulation and thereby assess the uncertainties of model output (Refsgaard et al., 2007). Out of four types of uncertainty i.e. statistical, scenario, qualitative and recognised uncertainty, MC simulation is capable of assessing statistical uncertainty (Refsgaard et al., 2007) that arises e.g. for historic and current activity levels, prices and experimental data on GHG emissions or reduction during the MACC exercise. MC estimates a range of possible model outcomes based on probability distribution functions[5] (pdf) that were previously defined for model variables or parameters. The convergence between the original model output space and the output space generated by MC simulation allows conclusions on the uncertainty of the model output. This can be particularly important to assess probabilities of model results exceeding a certain threshold e.g. for MACCs exceeding the carbon price threshold. MC approaches are well accepted in science for assessing model input uncertainty, but it can also help to identify model structure and projection uncertainty (Willows et al., 2003). MC simulation is a popular approach as it

[5] A Probability density function represents a certain probability that a model input or output shows a particular value. In case of continuous variation a pdf defines the probability that the value is between a certain range of values (Willows et al., 2003).

is applicable to many models, and various software tools are available for employing this technique. A disadvantage of MC simulation is that it cannot assess model context uncertainty and is strongly dependent on the selected pdfs for model inputs (Willows et al., 2003). This technique may also require large run times in case of complex models with a large number of input variables and parameters. There are several software packages available for MC simulation, add-ins for Excel spreadsheets including Lumenaut, ModelRisk, Quantum XL, Simtools amongst others. Modelrisk offers one of the largest set of analytical and reporting tools amongst the Excel based software and therefore this tool is used for the MC simulation in chapter 6. While describing the general methodology of MC, this section focuses on methodologies specific for Modelrisk Professional developed by Vose[6]. MC usually follows the methodological steps as stated below:

1) Identification of pdfs for model input variables and parameters
2) Model re-run based on a random number generation according to pre-defined pdfs
3) Combining the different possible model outcomes into one outcome range.

The identification of pdfs can be based either on data series e.g. historical data, expert opinion or probability distribution published in peer-reviewed publications. If historical data for input variables is available, there are various statistical techniques for automatically identifying the underlying probability distribution, including Maximum Likelihood Method, Methods of Moments and nonlinear optimisation. The Maximum Likelihood Method is probably the most popular method and is utilised by Modelrisk. This method requires independent and identically distributed values in a data series, which is often the case with historical data. It generates pdfs simply by setting values with a higher probability that occur more often and values with a lower probability which occur less often (for a detailed description see Law and Kelton, 2000). A larger dataset allows a better prediction. This method shows following advantages like a nearly zero bias of the input if the number of samples increases to infinity and it is a asymptotically efficient approach as no other unbiased estimation method has lower mean square error (Raychaudhuri, 2008). For published pdfs, the IPCC report is a good example as it states pdfs for EFs at Tier 1 or Tier 2 level. However, pdf identification is particularly problematic for ENG MACC assessment.

[6] The description of Modelrisk was partly obtained from http://www.vosesoftware.com/ModelRiskHelp.

ENG MACCs are based on interdisciplinary information with a large number of model variables and this makes the pdf identification a demanding task, not at least as the MACC developer may not be aware of all available data series or pdfs. Model inputs are commonly derived by expert judgement, and hence no pdfs or data series are available. This makes an automatic pdf derivation not possible. Data limitations for pdf generation within the MACC exercise are most noticeable for future projections, measure implementation costs and adoption rate and hence generation of pdfs must be derived by expert judgement. There are a vast number of different pdfs available. First, it is distinguished between discrete random variables i.e. representing finite number of distinct values and continuous random variables i.e. that can take infinite values. For discrete random variables, commonly applied pdfs include binomial distributions that describe the probability of occurrence of an event which are likely to happen or not, or Poisson distribution for small probability of occurrence but large number of observations (Goodarzi et al., 2013). For the MC applied in this study, the focus is entirely on continuous random variables. For these, the Gaussian distribution (or normal distribution) is probably the most commonly applied as it is a good approximation to many naturally occurring distributions e.g. population data; and with increasing size of the dataset, the normal distribution usually becomes more evident. The Gaussian distribution is most suitable for a small uncertainty range. It is defined by a mean that equals the peak of the distribution and the variance is given in a bell-shaped curve to the left and right of the mean (Figure 3.7).

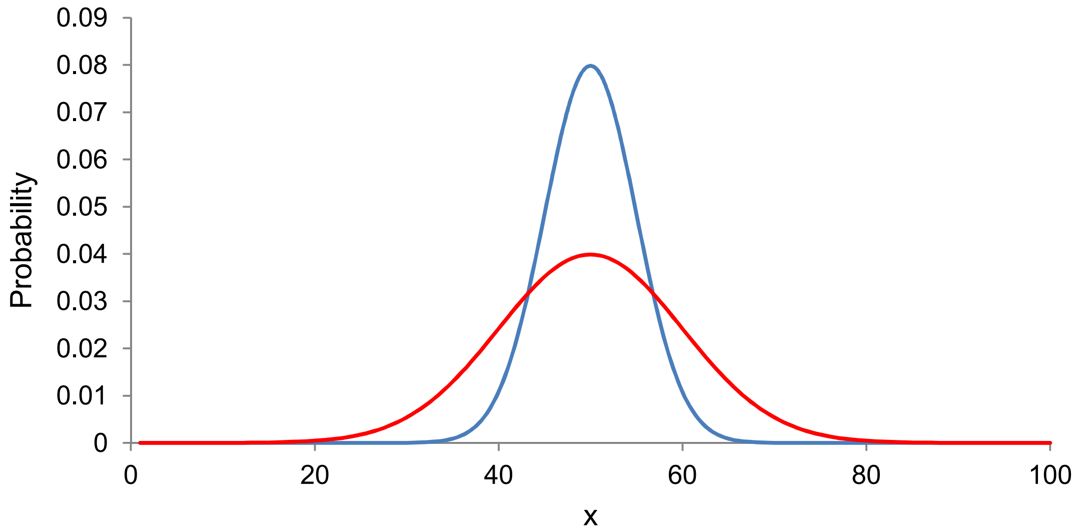

Figure 3.7: Example of a two Gaussian distributions.

Other available distributions that can be more accurate in describing a probability range include beta-, exponential-, gamma-, log-normal-, triangular or uniform distribution (Goodarzi et al., 2013). An uniform distribution allows choosing the values being between the minimum and maximum of that distribution at equal probability (Figure 3.8). It is usually used for very approximate uncertainty assessments with model input uncertainty not being understood well. Additionally, it may be good for representing expert opinion if an upper and lower bound is specified. For a MC simulation that compares the influences of all model inputs, it is problematic to assign to some variables a specific distribution and to others an uniform distribution. This is because the inequalities in treating uncertainty of model inputs will affect the results of MC negatively. Further, an uniform distribution is helpful to understand influences of variables and parameters on model output since the distribution assigns a broad range of possible values. Therefore, if the model output is insensitive to this range, the model will be even more insensitive to other distributions.

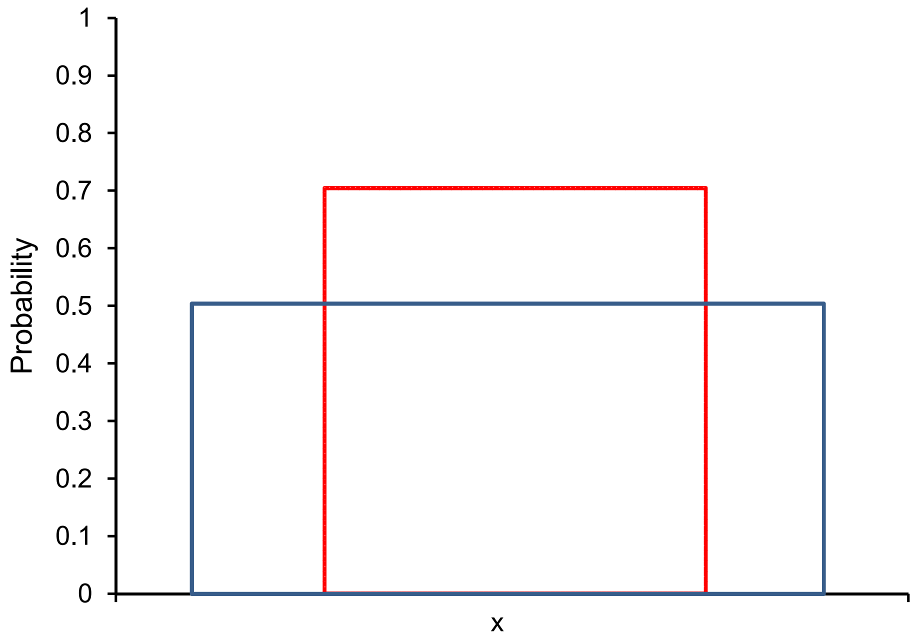

Figure 3.8: Example of two uniform distributions.

After defining and assigning the pdfs to the model inputs, MC simulation takes a random sample within the range of these pre-defined distributions. One of the earliest examples of random sampling is the experiment of Comte de Buffon in 1777, where a needle with the length L was randomly thrown on a horizontal plane with straight lines in the distance of d (d

> L) drawn on it (Kalos and Whitlock, 2008). The probability that the needle touches at least one of the lines was defined with $\frac{2L}{\pi d}$. A commonly applied methodology for random sampling, also called MC sampling, selects n values from a given dataset *N* with each value n having same probability to be chosen. Hence, the probability to select a value is *n/N* (Gentle, 2003). Generating random numbers requires a pseudorandom number generator (PRNG). Those numbers derived by software or hardware (and not by experiments) are called pseudorandom numbers since they are not truly random. Although most of PRNGs generate close to true random numbers, some PRNGs were criticised for e.g. correlation of consecutively followed numbers, no uniform distribution of large sequences and weak dimensional distribution of the output sequence. An early PRNG was developed by Neumann in 1946 where the generator selects the middle values of a squared original value in the interval [0,1] and squares this new value to continue with this procedure (Goodarzi et al., 2013). Modelrisk uses the 'Mersene Twister' PRNG for random number generation in the interval [0,1] following the MC sampling methodology. The algorithm was first published by Matsumoto and Nishimura (1998) and refined by Matsumoto and Nishimura (2002). It was well received for its ability to produce reliable random numbers and its improvements over previous generators in terms of number generation, speed and generation of very long numbers i.e. 32bit integers. However, the 'Mersene Twister' was criticised for its requirements of large computing power and for the initial version a long sequence of pre-runs was required to generate numbers that passes random tests (L'Ecuyer and Simard, 2007). Due to its general good performance, it is used for several software packages including MATLAB, Microsoft Visual C++ and SPSS. However, other sampling methods are also available e.g. the Bernoulli sampling where observations are selected independently with the sample size itself being a random variable or multistage sampling where primary values are selected first and thereafter values are chosen that are associated to the primary sampling unit (Gentle, 2003) and Latin Hypercube sampling (LHS; Iman et al., 1980). For latter the pre-defined pdf is split into n non-overlapping intervals that have equal probabilities of *1/N* to be selected according to *N* simulation runs (Refsgaard et al., 2007). LHS allows a more precise reflection of the pdf's shape compared to random sampling. Although LHS is commonly applied in risk analysis software, Modelrisk does not include this technique. Reasons could be that splitting the pdfs for each model input can increase simulation duration by multiple times depending on the number of input variables.

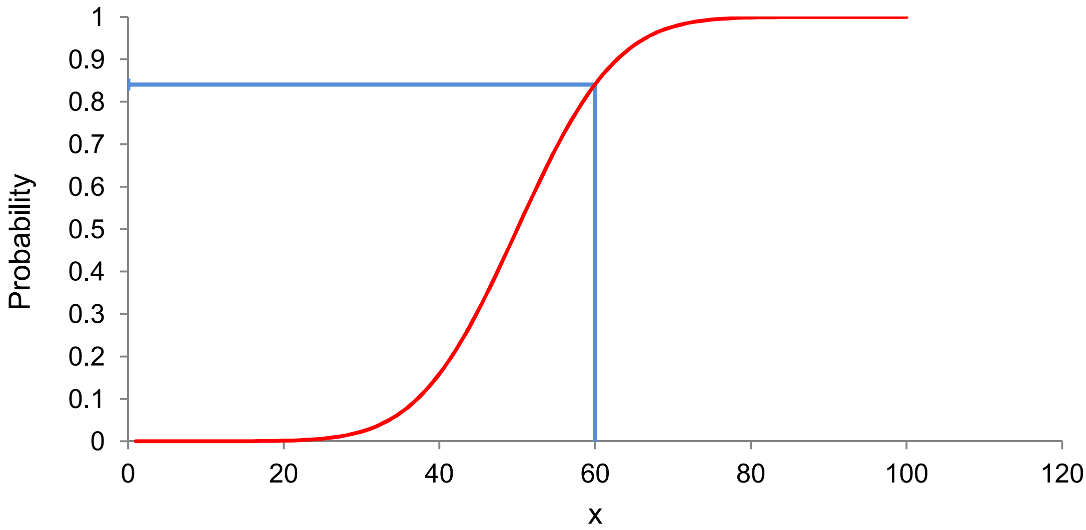

Figure 3.9: Example of random sampling with the inversion method to obtain the associated value in the cumulative density function. The generated pseudorandom number is 0.84 and the corresponding value is 60.

Based on the previously generated random numbers, Modelrisk and most other risk analysis models employ the inversion method to generate a random sample corresponding to a defined distribution. Hence, the random number (x-axis) is assigned to the corresponding value (y-axis) that represents a random value in the previously defined pdf or cumulative density function (cdf; Figure 3.9). Choosing from an uniform interval [0,1] ensures equal opportunity of a value being chosen in any percentile range of the distribution. Clearly, each pdf has its characteristics that define the cdf in terms of shape, scale and location. Other methods include cdf inverse method that generates first the inverse of a cdf and thereafter employs the same methodology as described above or Acceptance-Rejection method where the original pdf is substituted by appropriate pdfs (Gentle, 2003). Random sampling may however be subject to sampling errors in terms of pseudo-randomness if the sample size is limited. Therefore, the precision of the sampling results increases with the sample size. Finally, based on multiple transformations of the derived random numbers, the total value range for the model input is estimated. MC re-estimates the model outcome based on each generated random variable value multiple times, thereby defining a range of the model outcome.

Chapter 4 - Marginal abatement cost for the Chinese livestock sector

4.1 Introduction

The livestock sector is a significant GHG emission source in China. Compared to 1994, GHG emissions from enteric fermentation and manure management increased by 41% and 353% in 2005, respectively (NCCC, 2012 and 2004). This increase can be attributed to a massive increase of livestock production i.e. meat and milk production that increased between 1978 and 2012 by 8 and 16 times, respectively (OECD-FAO, 2014 and 2013). It is expected that in future, demand for livestock products per capita will continue to increase and this trend will be driven by rapidly increasing financial resources per capita with subsequent increasing affordability of livestock products. It is projected that meat demand per capita reaches OECD levels by 2022. To meet this demand, meat production must increase by 1.6% per annum (OECD-FAO, 2013) and thereby grain production by 2% per annum (Fan et al., 2012). Such an increase in livestock production levels remains an extremely ambitious target, particularly given the limitations unique to the agricultural sector in China (for further details see OECD, 2013):

1) Bio-physical limitations such as land availability are an issue in China, and the 12th five years plan set a cap on arable land use of 120 Mha mainly for restoration and conversion purposes. Land availability is further limited by increasing soil degradation in terms of salinisation, soil pollution and desertification (Akiyama and Kawamura, 2007). Latter being an urgent issue in the grasslands of northern China which harbours a large grazing animal husbandry industry.

2) Rising agricultural production costs that may lead to increasing imports from foreign subsidised agricultural production. Production costs are expected to rise due to increasing labour wages, scarcity of skilled and low wage labour in rural areas, increasing value of Chinese Yuan compared to USA Dollar or Euro and high

quality production methods that need to be adopted for meeting the increasing demand for high food quality.

3) Structural limitations are predominant since the Chinese agricultural sector consists mostly of smallholder farms e.g. with less than 0.1 ha cropland per household. These farms produce livestock goods inefficiently. Additionally, the farmer community is increasingly aging.

China's 12th five year plan focuses on these issues with priorities on rural poverty alleviation, food security including achieving self-sufficiency of food production, increasing agricultural productivity, improving water irrigation efficiency, promoting environmental sustainability and thereby restoring soil conditions, increasing political support and protection for agriculture and opening agricultural market for market-based trading (OECD-FAO, 2013). However, self-sufficiency coupled with more environmental friendly agricultural production can only be achieved with increasing production efficiency while simultaneously decreasing environmental pollution e.g. GHG emissions and depredation of natural resources (Shen et al., 2013). Although many advanced management techniques suitable for this purpose are available, research depicted mainly technical abatement potentials from mitigation options in Chinese agriculture (Hongmin et al., 2008; Huang and Tang, 2010; Lu et al., 2009; Nayak et al., 2015). However, focus on the economic mitigation potential is strongly under-represented. Despite its high direct and indirect contribution to total GHG emissions, the livestock sector gained far less attention in the climate change mitigation agenda in China than the arable sector. Identifying economic feasible mitigation in the Chinese livestock sector is strongly beneficial for policy makers, particularly in China as with availability of large budgetary resources, the financial support for the livestock sector increased strongly and could be used to support large-scale implementation of mitigation options. Identifying and implementing win-win technologies could further facilitate in achieving the ambitious targets set in the 12th five years' plan.

Rapid economic development, structural changes, abrupt governmental interventions and changes in food and taste preferences in China are some of the major constraints in accurately predicting future livestock activities. With the help of a SA, uncertainties attributed to future projections can be identified which is particularly important in China as assessment of alternative scenarios have been commonly weak in strategic environmental planning (Zhu et al., 2011). However, a SA would deliver valuable information for policy makers, particularly in a planned economy where future decisions on production and investment are embodied in a five year's plan. For a MACC exercise, most crucial input parameters that should be considered by a SA are output prices and animal inventories as they are subject to high uncertainty and strongly influence the assessment.

This chapter considers available information on the abatement potential for various mitigation measures and outlines the information sources used. The first section describes the information sources and parameterisation of the MACC and SA. The consecutive section shows the results of this exercise and thereafter discusses the findings in the context of the Chinese livestock sector including significance of findings for policy makers, institutional barriers for realising the abatement potentials and the importance of uncertainty assessment for decision processes.

4.2 Methodologies

Although the primary focus of this chapter is on the Chinese livestock sector, it also describes simplified methodologies and some findings from the assessment for the Chinese crop sector for a complete explanation of the full process of construction of this MACC. Further information on cropping activities can be read in Wang et al. (2014).

4.2.1 Projecting China's future agricultural activities

For this current MACC exercise, 2010 has been taken as the benchmark year. For obtaining agricultural activity data for the year 2010, historical data series for production levels, livestock number, cropping area and producer prices were gathered from the China Rural Statistical Yearbooks (MOA, 2001 - 2012a) and the China Livestock Yearbooks (MOA, 2001 - 2012b). For the initial MACC exercise, projections based on the CAPSiM model were utilised for generating a baseline scenario for livestock and cropping activities from 2010 – 2020 (in the following part of this chapter, the CAPSiM scenario will be referred to scenario I). It was presumed that the CAPSiM model projections were advantageous over other projections as it focussed specifically on China while including impacts of future political decisions and external factors such as agricultural production, consumption, trade and price development (IAASTD, 2009). Hence, it might provide consistent and robust projections in the Chinese agricultural context. CAPSiM results were supplied by the Centre for Chinese Agricultural Policy of the Chinese Academy of Science and access to this information is restricted to public. Table 4.1 and Table 4.2 show the past and predicted cropping area and livestock population.

Table 4.1: Past and predicted cropping area by CAPSiM model.

Cropping area(1000 ha)			
Crops	2010	2020	Annual change
Rice	29,873	25,612	-1.5%
Wheat	24,257	22,099	-0.9%
Maize	32,500	35,361	0.8%
Soybean	8,516	8,223	-0.3%
Cotton	4,849	5,168	0.6%
Oils	13,890	14,613	0.5%
Sugar	1,905	1,837	-0.4%
Total vegetable	19,000	19,040	0.0%
Greenhouse vegetable	3,553	3,560	0.0%
Open field vegetable	15,447	15,479	0.0%
Fruit	11,544	11,668	0.1%

Table 4.2: Past and predicted livestock numbers by CAPSiM model.

	Livestock population (1000 heads) †			
	Species	2010	2020	Annual change
Stock population	Non-dairy cattle	92,063	147,617	4.8%
	Dairy cattle	14,201	23,095	5.0%
	Sheep and goats	280,879	407,711	3.8%
	Horses‡	6,771	6,771	0.0%
	Asses‡	6,397	6,397	0.0%
	Mules‡	2,697	2,697	0.0%
Slaughter population†	Pigs	666,864	853,203	2.5%
	Poultry	11,005,780	14,297,441	2.7%
	Rabbits	454,455	740,259	5.0%

† Animal numbers from 2011 onwards are projected based on growth rates for product output (as reported by CAPSiM).
† Since majority of these species are only alive for part of the year, slaughtered animal number are reported.
‡ There are no projections for population of horses, asses, mules and rabbits available by CAPSiM. Based on historical development, it is assumed that population of horses, asses and mules remain stable, and rabbits grow by 5% per annum.
Source: altered from Wang et al. (2014)

4.2.2 Baseline GHG emissions

GHG emissions specific to cropping activities are reported in detail in Wang et al. (2014). A brief summary from this study is presented in the following. Specific EFs for China for indirect and direct N_2O emissions from cropland and manure were gathered from Gao et al. (2011), Wang et al. (2010) and Zhang et al. (2013), respectively. CH_4 emissions from rice paddies were based on the CH4MOD model (Zhang et al., 2011) and adjusted to projected rice cropping area in 2020. Manure production from Chinese livestock was estimated with regionalised manure production factors per animal species (Wang et al., 2006), while considering the share of liquid and solid compounds (Hang et al., 2012). EFs for manure management were applied based on Wang et al. (2010). CH_4 production based on enteric fermentation and manure storage were based on EFs reported by Fu and Yu (2010). Validation of the estimated livestock GHG emissions (e.g. from enteric fermentation and manure management) with GHG emission estimates from the Chinese national GHG inventory showed a high disparity. Therefore, it was assumed that GHG emissions reported by the GHG inventory for 2005 increased equal to the percentage increase estimated in this study.

4.2.3 Model farms for the Chinese livestock sector

Model farms were generated based on data from the China Agricultural Products Cost-Benefit Yearbook (NDRC, 1998 - 2012). We considered 6 different model farms separated by farm specialisation i.e. beef, dairy cow, sheep, goat, pig and poultry. These model farms were based on statistical averages. However, this approach cannot capture the overall heterogeneity of farms in terms of bio-physical and economical structures. While such a simplification allows understanding the full potential of mitigation options in a large country like China, it has to be clarified that these farms are hypothetical and abatement potentials and CE for single mitigation measures can vary for single farms in reality. While this research is an initial step in understanding the economic abatement potential in Chinese livestock production, further research must consider variations for real farms, ideally in small-scale assessments. Forecasts related to costs of production inputs including labour input, machine operations, fuel, electricity, technical service, material, maintenance, reparation, machinery use, feed e.g. concentrate or hay and animal purchase were based on historical development of these factors and expert judgement.

4.2.4 Parameterisation of mitigation options

For this MACC exercise, eight mitigation options were selected out of an initial long-list of 18 measures (Table 4.3). Following measures were excluded: i) grassland conversion to cropland, ii) cropland conversion to grassland, iii) afforestration of grassland, iv) reseeding of grassland, v) introduction of new grassland species, vi) feeding ionophores, vii) chemical defaunation, viii) feeding essential oils, ix) vaccination for reduced CH_4 production, x) concentrate feeding.

The parameterisation used for the MACC derivation is outlined in the following sub-chapters.

Table 4.3: Overview of mitigation options applicable to livestock and grassland.

No.	Measure	Target species
L1	Anaerobic digestion of manure	Cattle, dairy cows, pigs, poultry
L2	Breeding of livestock for increased yield	Indoor – cattle, dairy cows, pigs, sheep, goat
L3	Tea saponins addition to the diet	Indoor – cattle, dairy cows, sheep and goat
L4	Probiotics addition to the diet	Indoor – cattle, dairy cows, sheep and goat
L5	Lipid addition to the diet	Indoor – cattle, dairy cows, sheep and goat
L6	Grazing prohibition for 35% of grazed grasslands	Grazing – cattle, dairy cows, sheep and goats
L7	Reduction of stocking rate – medium grazing intensity	Grazing – cattle, dairy cows, sheep and goats
L8	Reduction of stocking rate – light grazing intensity	Grazing – cattle, dairy cows, sheep and goats

4.2.5 Abatement rates of mitigation options

For estimating the mitigation potential of the different mitigation options, abatement rates for each mitigation option that were specific to China were applied. For L1, the mitigation potentials for household anaerobic digesters as described in Zhang et al. (2012) were adopted in this exercise (Table 4.4). Measures L3 – L8 considered only ruminant and grazing species that are dominant in the Chinese livestock sector since measures L3 – L5 targeted CH_4 reduction from enteric fermentation and measures L6 – L8 targeted C sequestration in grazed grasslands (Table 4.3). Other species i.e. large herbivores and other non-ruminants (e.g. poultry) would only allow low mitigation potential for these mitigation options and were therefore not considered. Abatement rates for L2 - L5 were obtained from the MitiGate database and were specific to the Chinese livestock sector. The mean abatement potentials for L2, L3, L4 and L5 were based on 6, 29, 12 and 30 studies (Table 4.4). However, L2 and L4 showed an increase of enteric CH_4 for cattle (Table 4.4). This might be a puzzling result but reflects the findings of the Mitigate database. For L2, only six studies were included in the meta-analysis and only some of these focussed specifically yield increase. For L4, it was discussed in section 2.3 that probiotic addition to the diet shows strongly inconsistent results and these studies which focussed on cattle might have use microbial agents that did not decrease CH_4 production. However, these findings were considered in the present study and

L2 and L4 were not applied to cattle. It was assumed that L3-L5 were applied daily, and therefore only housed animals were considered for application. For measures L6 – L8, C sequestration potentials were obtained from the database of Nayak et al. (2015) which were also specific for Chinese conditions. For L6, L7 and L8, the abatement potentials were based on 44, 7 and 4 studies, respectively (Table 4.4).

Table 4.4: Abatement rates for the individual mitigation options.

Measure No.	Effects on yield increase	Abatement rate (per year)						
		Cattle (%/hd)	Dairy cow (%/hd)	Sheep (%/hd)	Goat (%/hd)	Mean (%/SU[†])	Grassland (tCO_2e/ ha)	Anaerobic digester (tCO_2e/ digester)
L1								2[†]
L2	1%	-11	6	8	8	4.1		
L3	5%	12	15	17	17	15.4		
L4	7%	-0.2	0.3	1	1	0.6		
L5	5%	8	16	14	14	14.3		
L6	1%						1.07	
L7	10%						0.7	
L8	3%						0.88	

[†] Sheep unit (SU) is a conversion factor that is used to compare the feed intake of different animal species. The conversion factor is for sheep, goat, cattle, dairy cow, pig is 1, 0.9, 5, 7 and 0.8, respectively.
Source: Wang et al., 2014

4.2.6 Measure implementation costs

On-farm implications and associated costs and benefits from measure implementation were identified through a literature review and expert consolidation. Table 4.5 shows the assumptions behind estimation of measure implementation costs. A social discount rate of 7% was applied to estimate 2010 values of the mitigation option costs.

Table 4.5: Variables and references for estimating implementation cost of mitigation options.

Option No.	Explications	Application rate	Major references
L1	The investment cost for an anaerobic digester at farm scale level is about ¥3250, but a subsidy between ¥800 and ¥1200 is provided by the government. The annual benefit of running a digester is estimated to be in average ¥500. It is assumed that one anaerobic digester is operational for 15 years and a relatively high failure rate of 8% of newly constructed digesters due to immense maintenance and technological shortcomings	Every 15 years	- MOA (2007) - NDRC (2007) - Zhang et al. (2012) - Han et al. (2008)
L2	Costs for high quality genetic material, artificial insemination and administration are ¥20, ¥40, and ¥20 per animal, respectively (the costs are adjusted to current prices). Due to the low success rate, sometimes more than one artificial insemination is done for one animal. The milk production and body weight will increase by 1% each year.	Annual	- Waldron et al. (2007) - Zhang and Beckman (2008) - expert judgement†
L3	A SU that is fed with 1g concentrated tea saponins per day shows increased milk production, body weight, and wool/cashmere production of 3%, 4%, and 4%, respectively. The feed intake increases by 2%. The purchasing costs for daily application are ¥1 per SU.*	Daily	- expert judgement†
L4	A SU that is fed with 1g probiotics per day shows increased milk production and body weight of 6%. The feed intake increases by 5%. The purchasing costs for daily application are ¥18 per SU.*	Daily	- Musa et al. (2009) - expert judgement†
L5	A SU that is fed with 40g polyunsaturated lipids per day shows increased milk production, body weight and wool/cashmere yield of 4%, 2%, and 2%, respectively. The purchasing costs for daily application are ¥219 per SU.*	Daily	- expert judgement†
L6 L7 L8	The cost assumptions for herders are based on farm surveys in Inner Mongolia. A model was generated that estimates the DM availability under different grazing intensities and hence the additional costs for supplementary feeding. Costs for machinery and labour input are based number of animals and area for hay making. The assumption was made that livestock is grazing freely which is most common in Chinese grazing systems. Thus, no additional costs apply for different grazing intensities. No construction of new warm sheds was assumed since the Chinese government increases the housing capacities strongly each year. Therefore, only costs regarding additional feeding and running housing facilities are applied. The increase of DM production ha^{-1} for L6, L7 and L8 is 1%, 10% and 3%, respectively.	Annual Annual Annual	- Farm questionnaires by the Inner Mongolia Agricultural University - Patton et al. (2007)

* Additional management costs of ¥2/animal apply for purchasing, transporting, feeding the feed additives.

† Since there is a gap in Chinese Scientific literature for the required information, several Chinese experts on their judgment of impact on yields and costs were consulted. The results presented here are the mean of all assumptions.

Source: Wang et al. (2014)

4.2.7 Adoption potential of mitigation measures

Adoption of mitigation measures during the baseline scenarios was obtained from either relevant policy targets set in China's 12th five years' plan e.g. for L1, L6, L7 and L8 or historical trends. The additional adoption potential during the mitigation scenarios were obtained from scientific literature and expert judgment (Table 4.6). For grassland area applicable to the mitigation options L6, L7 and L8, it was assumed that these are under heavy grazing pressure. Grassland utilisation rate is 50% and 35% for L7 and L8, respectively (Patton et al., 2007). It was assumed that mitigation options were adopted to their full adoption potential at a linear rate until 2020 i.e. 10% in 2011, 20% in 2020 and 100% in 2020.

Table 4.6: Application of mitigation options in baseline and mitigation scenarios.

Option No.	Historical or current adoption	Baseline adoption in 2020	Maximum feasible adoption in 2020*	References
L1	33% of total 120 M possible farm-scale anaerobic digesters	66% of total possible farm-scale anaerobic digesters	33% of total possible farm-scale anaerobic digesters	- MOA (2007) - NDRC (2007) - Zhang et al. (2012)
L2	Limited	Most common for beef and cow but practically non-existent for goat farms	20% of beef and dairy cattle, 30% of sheep, 60% for goat	- Waldron et al. (2007)
L3	Very limited	Very limited	10% of ruminant livestock since tea saponins are not sufficiently available	- Expert judgement
L4	10% of terrestrial livestock	Increasing adoption rate	50% of livestock	- Wang et al. (2008) - BSAC (2013)
L5	Limited	Limited	70% of livestock	- Expert judgement
L6	In 2010, 40% of Chinese grassland is under grazing ban, suspended grazing, or rotational grazing	60% of Chinese grassland is under grazing ban, suspended grazing, or rotational grazing	33% of grazing grassland	- MOA (2006) - MOEP (2005 - 2011) - Brown (2008)
L7	Limited	Limited	33% of grazing grassland	
L8	Limited	Limited	33% of grazing grassland	

* This is additional to the baseline adoption.
Source: Wang et al., 2014

4.2.8 Simultaneous implementation of mitigation options

The dietary (L3-L5) and grassland mitigation options (L6-L8) were applied mutually exclusive. For the dietary measures, an equal distribution of each feed additive was assumed to achieve 100% application for all livestock. Due to lack of more detailed data, it was assumed that the three different grassland measures are applied approximately to 1\3 of the total grazed grassland.

4.2.9 Scenario analysis - alternative scenarios

Following a modified methodology of Duinker and Greig (2007), key variables were first identified i.e. activity levels and agricultural prices as these are subject to high uncertainties and have a strong influence on the MAC assessment. As a second step, the study proposed to identify an uncertainty range of these variables and accordingly assign new values based on which future scenarios will be estimated. However, for this study it was not possible to re-estimate future scenarios for the Chinese agricultural sector based on these assumptions as there was no access to the CAPSiM model and its underlying assumptions and alternative models for re-estimation were not available. To overcome this limitation while simultaneously generating consistent scenarios, the following three alternative scenarios were used. Scenario II and III were based on well established models i.e. OECD-FAO agricultural outlook and the U.S. and world agricultural outlook by FAPRI, respectively. Scenario IV was based on extrapolation from historical data; here it was assumed that the development from 2010 to 2020 is equal to 2000 - 2010. These scenarios can be assigned to predictive forecast scenarios. Using this scenario type is justified for a cost-efficiency analysis (CEA) in near-term future as massive changes to the Chinese agricultural system are unlikely to happen. For longer projection periods, it may be beneficial to apply a broader range of possible scenarios. The selected scenarios included only a limited set of input variables of these that were required for the MACC exercise. In case of non-availability of certain forecast variables, it was assumed that these variables remained constant to the initial baseline scenario (scenario I). Production level projections were used to estimate yield changes in relation to varying livestock populations. For baseline GHG emissions, only those that were related to enteric fermentation and manure storage were re-estimated. Due to data restriction, it was not possible to segregate GHG emissions that arose from feed production for livestock. Additionally, the methodologies for crop sector assessment were specifically developed by the Chinese counterpart of this study. Cropping activities may not change significantly in response to varying demands from the livestock sector, given the strongly increasing import for animal feed. Relative to changing yields, manure excretion and enteric CH_4 production increased or declined and lead to consequently increasing or decreasing baseline GHG emissions. For measure cost estimations, model farms were adjusted to the new variables and measure implementation costs were re-estimated as described above.

4.3 Results

4.3.1 Baseline agricultural GHG emissions in China

Figure 4.1 shows the baseline GHG emissions from cropping and livestock activities and depicts an increase until 2020. Total agricultural GHG emissions are predicted to increase by 28.6% in 2020 as compared to 2010 levels i.e. a total GHG output of 1195 Mt CO_2e in 2020. GHG emissions from livestock (enteric fermentation and manure management) are 772 Mt CO_2e in 2020 i.e. an increase of 51% compared to 2005 levels (NCCC, 2012). This indicates that total increase of total agricultural GHG emissions is mainly driven by the livestock sector.

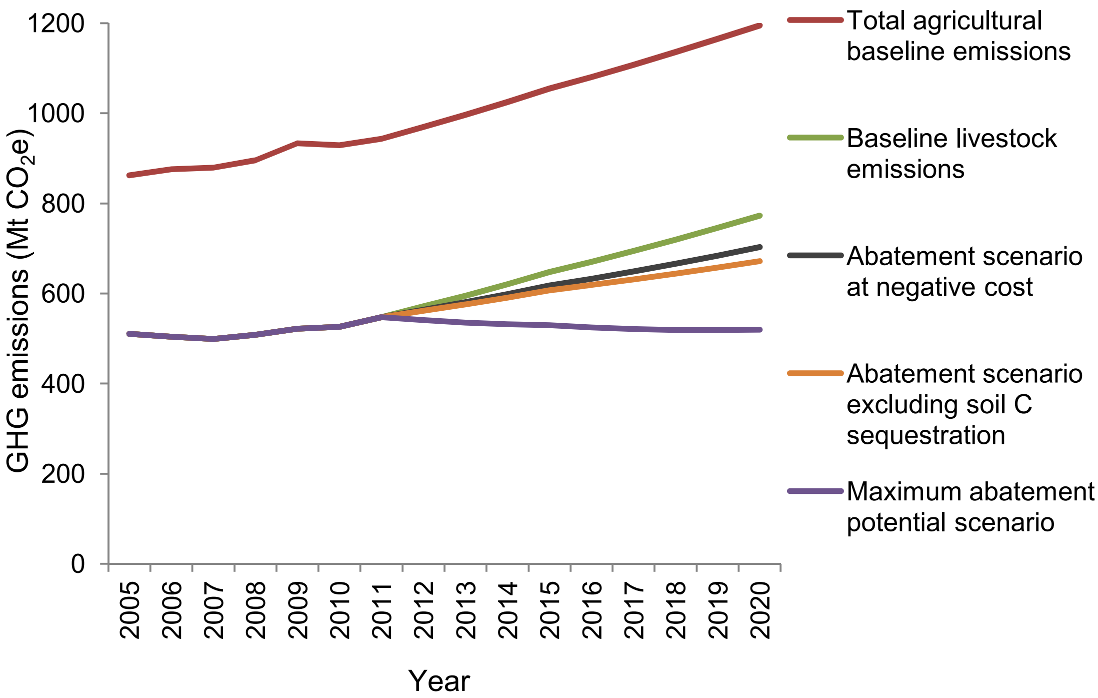

Figure 4.1: Baseline and abatement scenarios GHG emissions based on CAPSiM projections. Total agricultural baseline emissions are the cumulated soil N_2O emissions, rice CH_4 emissions, enteric CH_4 emissions, and manure management N_2O and CH_4 emissions. Baseline livestock emissions include only emissions from enteric fermentation and manure management. Abatement scenario at negative costs captures the abatement potentials of the negative cost measures L1, L2, L3 and L4. The abatement scenario potentials excluding soil C sequestration involves only these measures targeting CH_4 and N_2O emissions i.e. L1 - L5. The maximum abatement potential scenario combines the abatement potentials of all mitigation options.
Source: altered from Wang et al., 2014.

4.3.2 Mitigation potential and cost-effectiveness

The MACC shows a maximum abatement potential of 253 Mt CO_2e for 2020 for all mitigation options, representing a 34 % reduction of livestock baseline emissions (Figure 4.1 and Figure 4.2). When excluding the grassland measures, the abatement potential is 100 Mt CO_2e in 2020 (Figure 4.1 and Figure 4.2). There is a significant potential for win–win abatement. About 70 Mt CO_2e can be abated, while increasing income for the farmers which is equivalent to 9% of livestock baseline emissions in 2020. Thereby, resulting in cost savings of ¥ 23 billion. At a carbon price of ¥ 100 per tCO_2e, the MACC revealed that 110 Mt CO_2e or 14% of livestock baseline emissions can be realised.

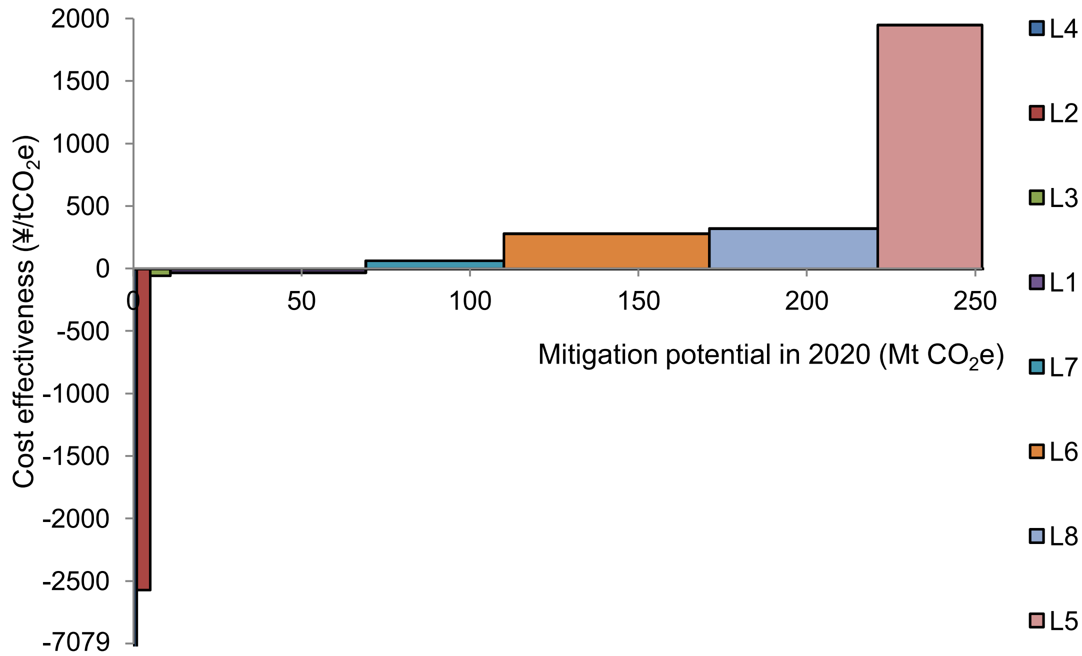

Figure 4.2 : MACC for the Chinese livestock sector in 2020 (discount rate = 7%). Measure code refer to measures in Table 4.3: L4 – Probiotics addition to the diet; L2 – Purebred breeding of livestock; L3 – Tea saponins addition to the diet; L1 – Anaerobic digestion of manure; L7 – Reduction of stocking rate – medium grazing intensity; L6 – Grazing prohibition for 35% of grazed grasslands; L8 - Reduction of stocking rate – light grazing intensity; L5 – Lipid addition to the diet.

Table 4.4 shows that abatement rates for tea saponin (L3) and lipid addition (L5) are highest in terms of CH_4 reduction from enteric fermentation by 15%. The grassland measures show large C sequestration potentials (Table 4.4). L1, L2, L3 and L4 are significant negative cost mitigation options; L1 shows further the highest GHG reduction potential amongst all livestock measures (Figure 4.2 and Table 4.7). The grassland measures also show impressive abatement potentials with L7 being favourite with an abatement potential of 40.77 Mt CO_2e available at low cost (Table 4.7). Although L5 shows an immense GHG reduction potential, this mitigation option is currently too expensive (Table 4.7). The most cost-beneficial measures are L4 and L2. The MACC results also show that political focus should be given to improved manure management systems.

Table 4.7 : Average abatement rate, cost, cost-effectiveness and abatement potential of mitigation options.

Option No.	Abatement rate (in 2020)		Cost (in 2020)		CE (in 2020)	Mitigation potential (in 2020)
	tCO_2e/ ha	CO_2e reduction/ SU^{-1} (in %)	¥/ha, 2010 price	¥/SU, 2010 price	¥/tCO_2e, 2010 price	Mt CO_2e
L1	2*		-500*		-32	58.66
L2		4.1		-29	-2571	4.4
L3		15.4		-3.4	-56	5.53
L4		0.6		-17	-7079	1.09
L5		14.3		109	1950	30.76
L6	1.067		300		281	60.78
L7	0.705		45		64	40.77
L8	0.877		283		322	50.72

* Per anaerobic digester

4.3.3 Alternative scenarios and their implications

The projections in the alternative scenarios vary strongly and sometimes with significant differences to scenario I (Table 4.8). For the modelled scenarios this is due to varying model capacity to predict the future and hence various aspects of social development are predicted differently e.g. economic growth, inflation or political change (Berkhout et al., 2002). For livestock population and production levels, scenario I shows the highest increase for all 5 animal categories i.e. beef cattle, dairy cattle, sheep and goats, pork and poultry with some exceptions e.g. dairy cattle population and production levels in scenario IV (Table 4.8). For producer prices, scenario I also shows amongst the highest increasing rates but other scenarios show much higher price increases i.e. for pork in all three alternative scenarios, beef cattle in scenario I, sheep and goats in scenario III and poultry in scenario IV (Table 4.8).

Table 4.8: Projections of animal stocks, production levels and producer prices in 2020 from different sources compared to 2010 levels (in %). Empty fields indicate non-availability of data.

Category	Scenario	Source	Beef cattle*	Dairy cattle†	Sheep and goats*	Pork*	Poultry*
Livestock inventory	I	CAPSiM	160,3%	162,6%	145,2%	127,9%	129,9%
	II	FAPRI	104,9%	123.7%[i]	85,8%	114.5%‡	
	III	OECD/FAO	107,2%	103,9%			
	IV	Historical†	86,0%	299,0%	100,5%	111.6%‡	117,2%
Production levels	I	CAPSiM	160,3%	153,6%	145,2%	127,9%	138,3%
	II	FAPRI	116,1%	162.5%[i]		125,5%	126.5%¯
	III	OECD/FAO	113,6%	118,2%	121,3%	116,4%	125,4%
	IV	Historical†	127,3%	432,1%	151,0%	127,9%	143,0%
Producer prices	I	CAPSiM	148,9%	164,1%	135,4%	139,0%	138,8%
	II	FAPRI	167,4%			178,9%	131,7%
	III	OECD/FAO	151,4%	114,8%	175,8%	246,9%	83,9%
	IV	Historical†	151,7%	124,8%		153,9%	176.1%±

* Main production value is meat from the respective species; production levels and producer prices address these commodities.
† The main production value is milk; production levels and producer prices address this commodity
† Based on official Chinese statistics from 2000 - 2010 (online accessible via USDA, China Agricultural and Economic Data).
[i] based on FAPRI 2011.
‡ Data only available for hogs.
¯ Data only available for broilers.
± Data only available for chicken.

The SA revealed that measures' CE and abatement potential was strongly affected by different scenarios. It is interesting that in all three alternative scenarios measures' abatement potential is lower compared to scenario I with largest differences for L1 in scenario II and L5 in scenarios II and III. This is also reflected in the total abatement potential that is 14%, 12% and 9% lower for scenario II, III and IV, respectively (Table 4.9). Compared to scenario I, abatement potential at negative costs is 90%, 21% and 19% lower for scenario II, III and IV, respectively. L1 remains cost negative for all scenarios except scenario II where it becomes cost positive; causing the significant lower abatement at negative costs compared to the other scenarios (Table 4.9). It is a striking result that the CE of L2 decreases strongly for all three alternative scenarios, thereby transforming it into a high cost measure that would generate at full application additional costs of ¥25.6, ¥96 and ¥17 billion (2010 price) in scenario II, III, IV, respectively (Table 4.9). The CE and abatement potential of grassland measures change only slightly within the different scenarios. The reason is that grazing livestock population remains stable in future and increasing livestock population will be kept indoor since pastures are currently at high grazing pressure, historical data does not show significant changes in grazing livestock population and there are strong political incentives to reduce the grazing pressure.

Table 4.9: Cost effectiveness and mitigation potential of the mitigation options under three alternative baseline scenarios

Option No.	Scenario I		Scenario II		Scenario III		Scenario IV	
	CE*	Mitigation potential†	CE*	Mitigation potential†	CE*	Mitigation potential†	CE*	Mitigation potential†
L1	-32	58.66	18	41.97	-1	46.95	-4	48.04
L2	-2571	4.4	9507	2.9	27071	3.53	5439	3.12
L3	-56	5.53	-218	3.7	-73	3.92	-362	4.67
L4	-7079	1.09	-12407	0.6	-7795	0.77	-17148	0.92
L5	1950	30.76	1722	22.57	2008	21.79	1454	26.00
L6	281	60.78	278	58.28	278	58.23	279	59.16
L7	64	40.77	60	39.09	50	39.05	54	39.68
L8	322	50.72	318	48.63	299	48.58	305	49.36
Total Mitigation†		252.71		217.74		222.82		230.95

* Cost effectiveness in ¥/tCO_2e at 2010 prices
†Mt CO_2e in 2020

4.4 Discussion and conclusion

Based on the published MACC for China (Wang et al., 2014), a SA was undertaken to visualise uncertainty involved in assumptions of future developments. To allow for assessment of other uncertainty sources, it was aimed to present the assumptions in a transparent manner.

4.4.1 Significance of the livestock sector

The initial MACC results highlighted the significance of the livestock sector as baseline GHG emissions are predicted to strongly increase by 2020. This sector will contribute significantly to total GHG emissions from Chinese agriculture in 2020. Therefore, it is of particular importance to implement climate change mitigation strategies in this sector. While Chinese livestock production is in a transition from small-scale to large scale indoor systems, policy maker should target biomass gasification (L1), breeding techniques (L2) and feed supplements as tea saponins (L3) and probiotics (L4) as these mitigation options can be best applied to housed livestock. L2, L3 and L4 can further increase productivity of livestock and could therefore deliver a solution to meet the rapidly increasing demand for livestock products in China (Jouany and Morgavi, 2007).

4.4.2 Negative and low-cost mitigation

The MACC results based on all four scenarios showed that a significant abatement potential is available at low or negative costs. Some reasons for the existence of these unrealised savings were discussed in chapter 3.3.1. However, there are some specific reasons attributed to the Chinese livestock systems. Herders prefer large herds over smaller herds which prevent a more efficient management of pastures (Wu et al., 2011). Further, the structure of Chinese livestock systems with millions of small-scale farms that constitute 90% of the sector can hinder implementation of more efficient management techniques since farmers may perceive opportunities for efficiency gains only as a small potential increase in income and may hence neglect these opportunities. However, with labour shortages in rural China, such efficiency gains are increasingly important as production levels could be maintained with reduced labour input. Difficulties in measure implementation also arise from the spatial wide distribution of these smallholder farms, low levels of mechanisation and occasionally no access to electricity; thereby leading to weak agricultural infrastructure and slow information dissemination as indicated by absence of extension advice in many regions. For instance, artificial insemination services for supplying high quality semen to farmers are poorly developed as smallholder farms show low purchase capacities and financially capable farms are spread over large regions. The challenge of implementing these cost-negative and low-cost mitigation options could be overcome by ambitious governmental investments in infrastructure.

4.4.3 Scenario analysis

Three of the four scenarios were institutional mid-term projections being based on capable and well resourced general equilibrium models with a broad number of components (Fu et al., 2012). Although these scenarios mostly matched in describing the trend of increase in livestock population, yield and producer prices, magnitudes of these trends were strongly different. This clearly shows the unpredictability of the future, particularly in the Chinese context. Policy makers should be aware of this prior to large scale implementation of mitigation options and carefully interpret the results of a CEA of the future. Considering only three key inputs i.e. livestock number, yield and producer prices could impressively depict the impact of different projections on abatement potential and associated costs. Breeding techniques (L2) turned from a cost negative to a high-cost mitigation option for all three alternative scenarios and are therefore under alternative conditions not advisable for large- scale implementation. This measure is only economically viable if high increase rates for livestock population, productivity and producer prices as anticipated by the CAPSiM scenario will be realised. Since the abatement potentials are in most cases lower as compared to the CAPSiM scenario, the policy maker should assume that estimates based on scenario I describe the upper end of potential GHG abatement. This is a significant result for the policy maker as the economic viability of the proposed mitigation options may change strongly if the future develops different than predicted.

Despite focusing only on the uncertainty source of future projections, this chapter could elaborate the significance of assessing the uncertainty of MACCs. It was intended to increase awareness of the important role of prediction uncertainty in a MAC assessment. It was shown that future MACC development should prioritise improvement of the accuracy of projections of the future. Scientists are responsible for doing so and report limitations of CEA's regarding the future. It can be concluded that before MACCs are utilised for policy decision support, these should undergo an uncertainty assessment to validate the findings in various future settings.

Chapter 5 - GHG emissions and technical mitigation potential of the European dairy sector

5.1 Introduction

Dairy production is of particular importance in Europe, being one of the largest producer, consumer and exporter of dairy products globally (Gerber et al., 2013a; Tacken et al., 2009) and with some of the most efficient production systems measured in terms of feed conversion (Krausmann et al., 2008; O'Mara, 2011). Despite this, further improvement in emission intensity is likely to be needed to meet increasingly binding environmental targets. In 2004, the European livestock sector emitted 623 Mt CO_2e with 28-30% of total attributable to dairy production, which contributed 40% to total CH_4 emissions emitted by enteric fermentation and anaerobic digestion of manure during storage (Weiss and Leip, 2012).

Currently there are no direct environmental regulations enforcing any emissions reduction in European dairy production. But several European regulations can have indirect impacts on dairy related GHG emissions. For instance, the Nitrates Directive (91/676/EEC) limits N input to soils in NVZs and hence lowers soil N_2O emissions. Moreover, the European Commission supports organic and low-input farming practices that reduces GHG emissions from dairy farms, due to lower production intensity (Olesen et al., 2006). But indirect approaches are likely to be insufficient to reduce the sectors' GHG emissions significantly and it is important that clear mitigation measures are identified to guide more direct policy interventions. Specifically there is a need to understand the suite of technical measures and their technical abatement potentials. This is a necessary and preliminary step for assessing the economic abatement potential in the European dairy sector as only technical feasible mitigation options can qualify for a MACC.

Based on the findings of this chapter, a MACC for the EU-15 dairy sector will be developed. While the assessment of baseline GHG emissions and reduction potentials in this chapter are based on sophisticated and European-wide top-down models, it is an innovative approach to combine these estimates in an ENG MACC. For Europe is a considerably larger dataset available e.g. on activity levels, bio-physical conditions, management techniques and impact of mitigation options compared to China. This information sources should be utilised to draw projections on in-depth assessment that will improve the certainty of the abatement potentials estimated for dairy production at a large scale. This is particularly important as the structure and activity levels of the dairy sector will change drastically due to the abolishment of the milk quota and increasing global demand for dairy products; thus consequences for GHG baseline emissions and abatement must be understood and this research contributes to the scientific understanding.

This chapter will estimate technical abatement potentials in the EU-15 dairy sector in 2020. There is no study assessing this specifically for the EU-15 dairy sector. New EU member countries were excluded since production intensity and mechanisation levels vary considerably between new and older member countries and this may hinder measure implementation. Further, dairy production levels in new member countries are low compared to EU-15**.** The first section explains the modelling approach for this study drawing on existing modelling methods. The result section shows the 'business as usual' (BAU) development of the dairy sector until 2020 and based on this total and per unit product GHG emissions in 2008 and 2020 were estimated. Thereafter, the technical abatement potential of nine mitigation options was assessed. The final section discusses issues with intensifying dairy production and how the mitigation options can contribute to further intensification. The final section highlights the limitations of the study design with a focus on uncertainties that may arise.

5.2 Methodologies

5.2.1 Modelling approach

The GHG reduction potentials of 9 mitigation options were estimated for the benchmark year 2008 and the final projection year 2020 with the MITERRA-Europe model. Similar to the MACC approach, baseline GHG emissions were first estimated to be followed by technical abatement potential for each mitigation option. The following GHG emissions were considered: i) direct emissions from dairy production including young cows (enteric fermentation and manure storage) and ii) indirect emissions from total feed production for the dairy sector within EU-15 (N_2O soil emissions and SOC changes), on-farm fuel and electricity use and fertiliser production. GHG emissions from product processing, transport, arable land and LUC outside Europe i.e. for concentrate production were excluded. Here, LUC for soybean production in South-America is likely to have a substantial contribution to the sector's GHG emissions (Flysjö et al., 2012). Interaction of measures was not accounted for.

5.2.2 MITERRA-Europe

Wageningen UR developed the MITERRA-Europe model as part of a contract with the European Commission (Velthof et al., 2007). Initially, the model was designed to assess the impact of reducing N input in European agricultural systems on the environment in terms of water and air pollution and GHG emissions; thereby enabling to quantify the impacts of policies in the European agricultural sector (Velthof et al., 2007). Since then the model was further improved and focussed more strongly on GHG emissions from cropping activities (Velthof et al., 2009). The model can assess annual N_2O, NH_3, NO_x, NO_3, CO_2, and CH_4 emissions in European agriculture in a deterministic approach [7] at NUTS-2 level (Nomenclature of Territorial Units for Statistics; Lesschen et al., 2011). MITERRA-Europe includes a N leaching module, SOC module and a module for mitigation options that can reduce GHG emissions. For a detailed description of all inputs and calculation methodologies see Velthof et al. (2009) and Lesschen et al. (2011). MITERRA-Europe was partly based on model inputs from the CAPRI model, Greenhouse Gas and Air Pollution Interactions and Synergies model (GAINS) and various other inputs e.g. from Eurostat and FAO.

CAPRI (www.capri-model.org) is a model for the EU-27 agricultural sector at NUTS-2 level considering agricultural supply from 35 crops and 19 animal categories (Lesschen et al., 2011). Model outcomes included cropping area, animal number, environmental indicators and costs of agricultural and environmental policies. CAPRI also projected future agricultural activities. This includes also increasing crop and grass yields which are determined exogenously by trend analysis of data from EUROSTAT (Britz and Wiske, 2008; Table 5.1). A detailed description of the CAPRI modelling system can be found in Britz and Witzke (2008) and Leip et al. (2010). The GAINS model (www.gains.iiasa.ac.at/gains/) estimated N and C emissions in Europe from agricultural and other sectors, and includes detailed databases on agricultural activities for the present and future (Lesschen et al., 2011).

[7] A deterministic model is a model that results in a determined outcome without effect of random variation.

The MITERRA-Europe model was based on inputs from various models and other data sources. All model input parameters can be summarised in four main categories: activity levels, biophysical framework, EFs and GHG reduction per ha or animal of the mitigation options (Figure 5.1). Dairy cow numbers and land area for feed production and grazing were important inputs for MITERRA-Europe. Animal numbers were obtained from GAINS at national level and distributed to NUTS-2 level according to CAPRI. Based on the feeding regime and crop yield, total cropland and grassland were estimated that was required to meet the energy and nutrition demand from dairy cows (Table 5.1).

Table 5.1: Overview of crop- and grassland area with specific yield and fertiliser input in the EU-15 dairy sector for 2008 and 2020 (EU-15 averages).

	Year	Fodder maize	Feed cereals	Other fodder	Permanent grassland	Temporary grassland	Rough grazing
Area (M ha)	2008	2.49	2.08	1.74	15.54	6.26	5.49
	2020	2.49	2.06	1.66	15.56	5.97	5.49
Yield (t/ha)*	2008	39.9	6.2	25.3	5.1	25.3	
	2020	47.8	7.0	28.1	5.1	28.1	
Fertiliser input (kg N/ha)	2008	109.8	88.0	109.3	50.6	109.3	
	2020	133.1	91.6	113.9	47.3	113.9	

* Crop yield expressed in fresh matter; permanent grassland yield expressed in dry matter

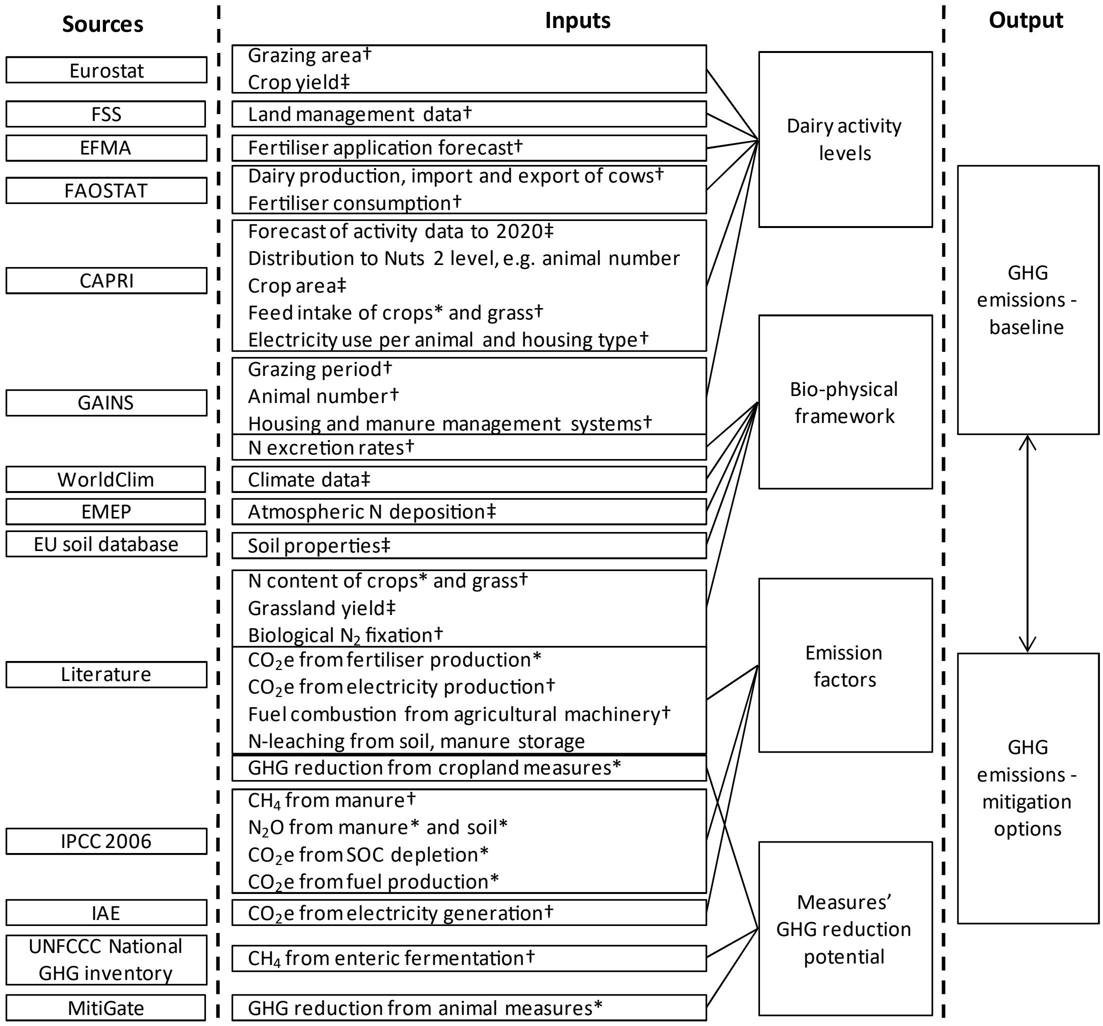

Figure 5.1: Conceptual model including data input of MITERRA-Europe and desired model output. Spatial scales of the inputs are: * European level; † country level; ‡ regional level. The abbreviations in the source section are as follows, FSS: Farm Structure Survey; EFMA: European Fertiliser Manufacturers Association; CAPRI: Common Agricultural Policy Regionalised Impact analysis; GAINS: Greenhouse Gas and Air Pollution Interactions and Synergies-Model; WorldClim: WorldClim Global Climate Data; EMEP: The European Monitoring and Evaluation Programme; IAE: International Energy Agency.
Source: altered from Lesschen et al. (2011), Velthof et al. (2009) and own source

The feeding regime was based on CAPRI data for the year 2004 and for most countries the feed intake per cow increased until 2020, based on higher feed requirements of more productive dairy cows (Figure 5.2). Land use in Europe related to the dairy sector included cultivation of fodder maize, other fodder (e.g. Lucerne), feed cereals and three grassland types i.e. permanent grassland, temporary grassland and rough grazing (grassland areas on difficult accessible areas; Table 5.1; Velthof et al., 2009). Based on Eurostat data the share of the three grassland types was estimated. Grassland and crop yields are based on Smit et al. (2008) and FAOSTAT data, respectively (Figure 5.1 and Table 5.1).

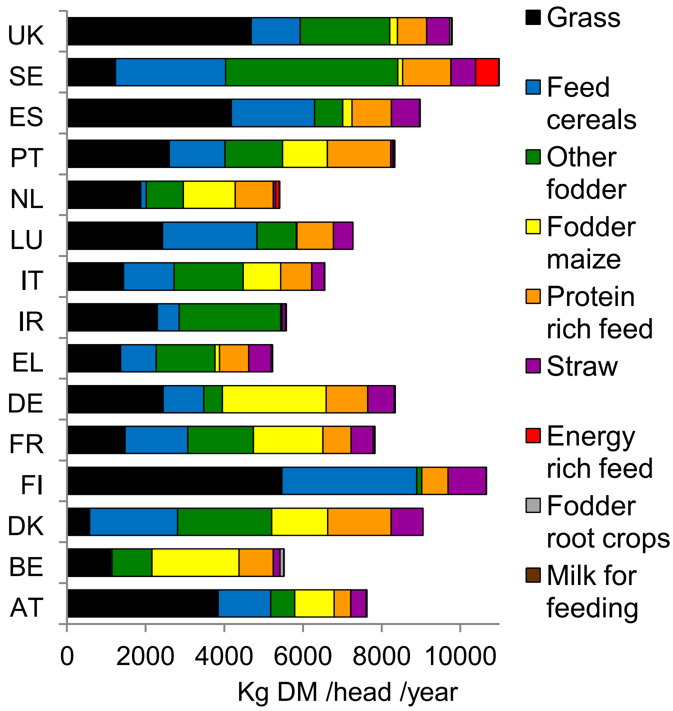

Figure 5.2: Dairy cows' feed intake per EU-15 member country in 2020. The country code translates as follows, AT: Austria; BL: Belgium; DE: Germany; DK: Denmark; EL: Greece; ES: Spain; FI: Finland; FR: France; IR: Ireland; IT: Italy; LU: Luxembourg; NL: Netherlands; PT: Portugal; SE: Sweden; UK: United Kingdom.

5.2.3 Baseline projection of the EU-15 dairy sector until 2020

For detailed information on dairy sector's activity levels until 2020, CAPRI projections were included for animal number, area for crops and grassland, crop yields, feed intake and manure production (Table 5.1 and Table 5.2 Figure 5.2). Dairy cow stock was predicted to reach 15 M head in 2020, a 17% decrease from 2008 levels (Table 5.2). Milk yield was projected to increase annually by 1.5% from 2008 to 2020, while taking the abolishment of the milk quota into account (DG-AGRI projections); thereby exceeding production levels of 2008 by 6% and indicating an increase in production efficiency across EU-15 (Table 5.2).

The DG-AGRI projections stated several reasons for this development as genetic improvement, utilisation of robots, improved management of pastures and higher levels of concentration in the diet. Main drivers are the increasing global demand for dairy products, particularly from China. However, milk yield increase based on genetic improvements may have reached its limits in some member countries as for instance reduced fertility of high producing cows is an issue of concern (Dobsen et al., 2007).

Table 5.2: Overview of dairy cows and their milk yield in 2008 and 2020 per EU-15 member country.

Country	2008			2020		
	Dairy cows (M head)	Young cattle (M head)	Milk (t/cow)	Dairy cows (M head)	Young cattle (M head)	Milk (t/cow)
AT	0.53	0.8	6.1	0.46	0.59	7.3
BL	0.52	1.04	5.5	0.55	0.83	6.6
DE	4.23	5.71	6.8	3.48	3.09	8.1
DK	0.57	0.79	8.5	0.49	0.56	10.1
EL	0.15	0.24	3.8	0.14	0.21	4.5
ES	0.89	2.8	7.1	0.77	2.45	8.6
FI	0.29	0.44	8	0.21	0.29	9.6
FR	3.8	7.45	6.5	3.21	6.37	7.8
IR	1.09	2.79	4.8	1.15	2.14	5.8
IT	1.83	2.62	6.1	1.65	2.02	7.4
LU	0.05	0.08	6.9	0.03	0.06	8.3
NL	1.59	1.25	7.7	1.73	1.24	9.2
PT	0.3	0.48	6.4	0.24	0.35	7.7
SE	0.37	0.72	8.3	0.31	0.41	9.9
UK	1.9	4.82	7.2	1.63	4.34	8.6
EU-15	18.11	32.01	6.6*	15.05	24.95	8*

* EU-15 weighted average

Highest dairy cow density per ha of utilised agricultural area (UAA) was in Netherlands, Belgium, Germany and parts of Italy, Greece and Finland in 2008 and 2020; with a decreasing density across Europe by 2020, except for some regions in Spain, Italy and the United Kingdom (Figure 5.3). Despite an increase in the DMI of dairy cows by 17%, the UAA is predicted to decrease by 1% in 2020 as compared to 2008. Increased feed demand in

2020 was met by more productive crop- and grasslands (Table 5.1). Projections for inorganic fertiliser use were based on projections from Fertilizers Europe and were predicted to be stable until 2020 (Table 5.1; www.fertilizereurope.com). The increase in fertiliser input is based on a higher application of organic manure as more productive cows produce more excreta. A linear annual change of activity levels from the MITERRA-Europe results in 2008 to the projections in 2020 were assumed.

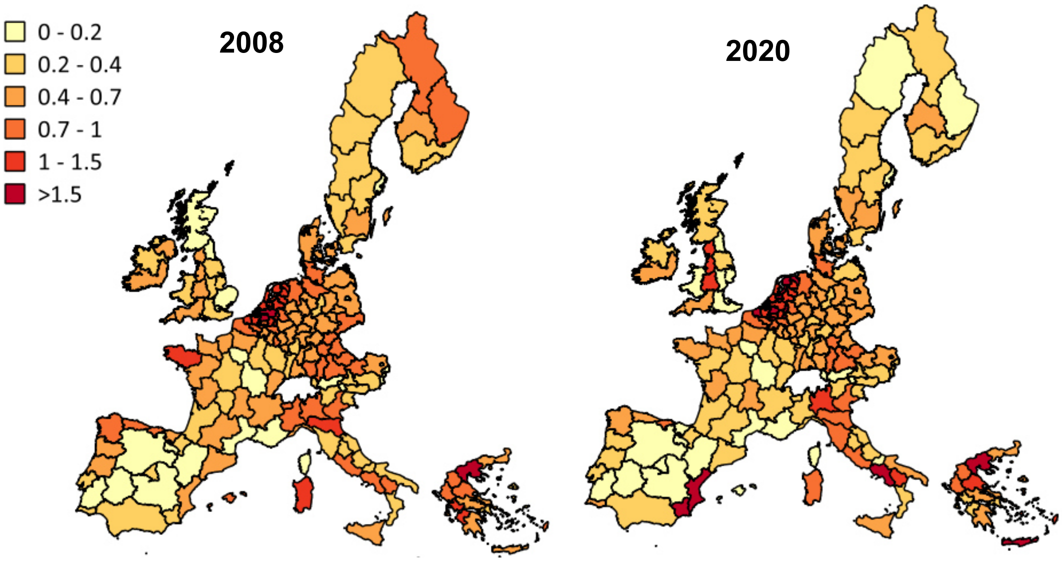

Figure 5.3: Dairy cow density in 2008 and 2020 in EU-15 (head/ ha utilised agricultural area).

5.2.4 Baseline GHG estimation of EU-15 dairy sector

CH_4 emissions from enteric fermentation were estimated using EFs from national GHG inventories (data from 2010). CH_4 and N_2O emissions from manure management were based on IPCC EFs (Eggleston et al., 2006), where the first was dependent on annual average temperature and manure storage system in the targeted country, and the latter was multiplied by country specific N-excretion rates (Figure 5.1; Lesschen et al., 2011).

Soil N_2O emissions were based on IPCC EFs, except for N leaching, since MITERRA-Europe uses its specific methodology (Lesschen et al., 2011). All N inputs to crop- and grasslands were accounted i.e. from fertiliser and manure application, crop residuals, urine and faeces. Since no data on manure application was available, it was assumed that all manure outside NVZs was applied to crop- and grasslands, whereas inside NVZs, only 170 kg N/ha was applied, or higher for some countries in case of derogations (Velthof et al., 2009). In case of manure N exceeding these limitations the manure was transported to other NUTS 2 regions in the specific country and if this excess applied for the whole country the remaining manure is treated and removed from the sector (Velthof et al., 2007).

Changes in SOC were specific for arable- and grasslands and were based on the stock change factor approach as described in the IPCC guidelines (Eggleston et al., 2006). The change of C stock was the difference between the C stock in 2008 and 2020 while the C stock for both years was estimated by integrating changes in land use, soil management and C inputs to a reference C stock which is climate and soil type specific (de Wit et al., 2014). Additionally, peat oxidation by managing organic soils e.g. tillage were accounted for. These and CO_2e emissions from liming and urea application were based on IPCC EFs (Lesschen et al., 2011). CO_2 and N_2O emissions from N fertiliser production based on EFs from Brentrup and Pallière (2008) were accounted for.

Average fuel consumption for cultivation of different crop species and grasslands was based on Lesschen et al. (2011). This study considered varying mechanisation levels and hence fuel consumption efficiency across Europe. CO_2e emissions from production per litre (L) of diesel were based on Eggleston et al. (2006). Electricity consumption for animal housing were derived from CAPRI while countries' average annual temperature determined heating and hence electricity requirements (Lesschen et al., 2011). The country specific EFs from electricity production were based on the International Energy Agency (IEA, 2010).

5.2.5 Mitigation options for the EU-15 dairy sector

Nine mitigation options were identified for the EU-15 dairy sector where at least two mitigation options focus on the main mitigation opportunities in livestock production i.e. enteric fermentation, soil N_2O emission and soil C sequestration (Table 5.3). The mitigation options could be categorised as: i) applied on agricultural land i.e. optimal fertilisation (M1), reduced tillage (M2), cover crops (M3) and reduced fertilisation during winter (M4); and ii) related to the animal i.e. feed supplementation with lipids (M5), nitrate (M6), tannins (M7) and probiotics (M8) and animal genetic selection (M9). Although several measures were available for manure management (Petersen et al., 2013) with anaerobic digester showing high GHG reduction potentials (Gerber et al., 2013a), these measures were excluded, as manure management systems vary strongly across EU-15 and only limited data on manure systems was available.

5.2.6 Abatement potential of mitigation options

5.2.6.1 Agricultural land measures

For the agricultural measures, M1 and M4, the methodology from Velthof et al. (2009) were adopted to estimate GHG reduction potentials with MITERRA-Europe (Table 5.3). For M2 and M3, the reduction potentials were assessed based on a literature review. Reduced tillage was assumed to increase SOC sequestration and to reduce GHG emissions from fuel combustion due to less intensive ploughing practices. For M3, decreasing soil N_2O in intensive agricultural systems, due to lower fertilizer use and less nitrate leaching, and increasing SOC sequestration were assumed (Table 5.3).

5.2.6.2 Animal measures

For the dietary mitigation measures (M5-M8), GHG reduction potentials were used based on MitiGate and Lewis et al. (2013); these were verified by expert consultation. In case of strongly divergent results between these information sources, a mean between both was assumed.

For M5, enteric CH_4 reduction potential was reported by 11 studies for dairy cows in the Europe and USA (Mitigate; Table 5.3). For M6, only 1% of DMI will be supplemented by nitrates to avoid possible toxic effects. The MitiGate database reported 29% enteric CH_4 reduction by supplementing animals' diet with 2.2% nitrate (based on one study for dairy cows in Europe). Assuming a linear response, the expected enteric CH_4 reduction potential is 13% (Table 5.3). For M7, tannins extracted from chestnut wood are fed to the cows. The MitiGate database showed an enteric CH_4 reduction by 1% (based on 11 studies for dairy cows in Europe and USA) but Lewis et al. (2013) showed a mean reduction potential of 15%. This divergence is due to the type of tannins used (synthetic tannins, tannin extracts or tannin rich forage) in the studies. The mean between both meta-analyses was assumed (Table 5.3). For M8, Lewis et al. (2013) show strongly inconsistent results of the effect of probiotics, mainly due to varying effect of different microbiological agents. Therefore, reduction potentials from the MitiGate database were used (based on 14 studies for all ruminants in Europe and USA; Table 5.3). M9 includes selection of animals for the traits of reduced enteric CH_4 output and increased milk yield. Currently, there is no literature available on possible reduction potentials; hence the GHG reduction potential and milk increase were based on expert judgement (Table 5.3).

5.2.7 Baseline activity of mitigation options

GHG reduction of mitigation options was estimated from measure's adoption that is additional to measure's application during the BAU scenario. For adoption potential of M1, the MITERRA-Europe specific methodology was used (Table 5.3). Fertiliser application slightly increases in 2020, but crop yield and hence crop N demand also in the meantime increases strongly. This leads to a reduced application potential of M1 in future (Table 5.3). For M2 and M3, statistical data on cropland under conventional tillage were gathered from the survey on agricultural production methods (SAPM) and area under winter crops were obtained from the farm structure survey (FSS; Table 5.3). In 2010, the FSS was a complete agricultural census including all agricultural holdings with at least one ha of UAA and included the SAPM that gathered information on agricultural management techniques (available at Eurostat). However, data for M2 and M3 was only available for total European agricultural sector and hence the share of these land management systems from the dairy sector's UAA was assumed to be the same as that from total UAA.

For BAU adoption of M5, M6, M7, M8 and M9, there was no statistical data available. Assumptions were based on scientific literature and expert judgment. M5 and M8 became increasingly popular amongst livestock farmers since they delivered yield improvement as an appropriate alternative to antibiotics that are banned in European livestock production (Franz et al., 2010; Grainger and Beauchemin, 2011; Table 5.3). In 2014, M6 was not available in Europe and application showed no yield increase and risk of intoxication by applying an overdose. Hence, no BAU implementation of this measure was assumed (Table 5.3). Also for M7 there is a low measure adoption during the baseline since extracted tannins are costly, and overdose can affect digestibility and DMI negatively. However, tannins have a positive impact on milk yield (Table 5.3). Animal selection is a common practice in European dairy production with nearly 100% application in North and West Europe and substantially lower in South Europe (van Arendonk and Bijma, 2003). It was expected that only a few farmers will change their breeding scheme to opt for reduced enteric CH_4 production without external forces. Therefore, this measure is mainly projected to be applied by farmers without a breeding scheme in 2020 (Table 5.3).

Table 5.3: Parameterisation of mitigation options in MITERRA-Europe and their activity levels in the 2020 baseline.

Measure	Parameter	Parameter description
M1 - Optimal fertilisation	MITERRA	N inputs to soils i.e. manure and mineral fertiliser application, N excretion during grazing, atmospheric N deposition, biologically fixed N and gross N mineralisation were summed up and corrected by loss from run-off and mineralisation to estimate total soil N. After, mineral fertiliser input was adjusted so that soil N is equal to crop N demand (i.e. N content of the harvested crop and crop residue corrected by the crop N uptake factor).
	Activity level	Based on crop N demand and total soil N input 2020, the divergence to an optimal fertilisation rate was estimated. In some countries, this could result in a reduction of mineral N fertiliser if total N input was higher than N plant demand.
M2 - Reduced tillage	MITERRA	Reduced tillage increased C stocks by 2-8% (de Wit et al., 2014) and reduced CO_2e from fuel consumption by 0.016t CO_2e /ha /year (King et al., 2004). No changes in N_2O emissions were assumed since scientific literature was not consistent.
	Activity level	Reduced tillage was only applied to cropland under conventional tillage since application on non-tilled land increased SOC depletion. Conventional tilled area did not change from 2008 in 2020 baseline as no driving forces were expected.
M3 - Cover crops	MITERRA	After ploughing cover crops into the soil, fertiliser application was reduced by 25% in intensive agriculture (N-surplus of 100 kg N /ha) and not reduced in less intensive systems. Cover crops increased N uptake from crops and reduced N runoff and leaching by 25% (Velthof et al., 2009). The C input is increased from medium to high und thus enhanced C sequestration rate.
	Activity level	Cover crops were applied to all croplands during the fallow period without winter crops. The applicable area was up to 80% in warm and 25% in cold regions (average annual temperature <10°C) but corrected for a share of winter crops. Cover crop area increased annually by 2% until 2020.
M4 - Restricted fertilisation during winter	MITERRA	25% of manure production was assumed to be applied to arable land during winter i.e. with no N uptake from crops. This measure prohibited manure fertilisation during that season and consequent losses. The additional manure was applied to the soil in spring and hence less mineral N fertiliser was required. 50% of N in manure was available to the plant (Velthof et al., 2009).
	Activity level	This measure was fully applicable only outside NVZs and 20% within NVZs since it was assumed that some farmers applied manure in winter despite prohibition. The NVZ per country did not change until 2020.

Table 5.3: continued

Measure	Parameter	Parameter description
M5 - Dietary lipids	MITERRA	There was no data on diets' lipid content available. A lipid addition only by 2.5% of DMI was assumed to prevent for overdosage. Thereby, 13% of enteric CH_4 production was reduced, and milk yield was increased by 5%.
	Activity level	It was assumed that 30% of dairy cows were be fed with additional fat sources in 2020. The remaining cows could be fed with dietary lipids.
M6 - Dietary nitrate	MITERRA	By supplementing 1% of DMI with nitrate enteric CH_4 production was reduced by 13%. Nitrate was only fed to adult dairy cows and did no effect N excretion rate since nitrogen-rich feed (grass silage) was replaced by nitrate and hence N content in the diet stayed balanced.
	Activity level	No baseline application of nitrate feed supplementation was assumed.
M7 - Dietary tannins	MITERRA	Extracted tannins were be added at a level of 1% of DMI; thereby reducing enteric CH_4 by 7% and increasing milk yield by 5%. Tannin addition did not affect N excretion rate of dairy cows.
	Activity level	Tannins were rarely applied in dairy production in 2008. Therefore, a low baseline application by only 10% of dairy farmers in 2020 was assumed.
M8 - Dietary probiotics	MITERRA	Probiotics were be fed in a dose of 10 g/day/animal. Enteric CH_4 production thereby reduced by 3% and milk yield increased by 3%. Probiotics were only fed to adult dairy cows.
	Activity level	It was assumed that 30% of dairy cows are fed with probiotics in 2020.
M9 - Animal selection	MITERRA	Selection for reduced enteric CH_4 emission and increased milk yield reduced 9.5% of enteric CH_4 production and increased milk yield by 10% in 2020 (Haas et al., 2011). A linear adoption until 2020 was expected.
	Activity level	This mitigation option applied to 10% of dairy cows in North and West Europe and 25% in South Europe in 2020.

5.3 Results

5.3.1 Baseline GHG emissions in 2008 and 2020

Total GHG emissions (excluding SOC emissions) are estimated to be 166 Mt CO_2e in 2020, an 8% decrease from 2008 levels; and only the Netherlands shows increasing total GHG emissions (Figure 5.4).

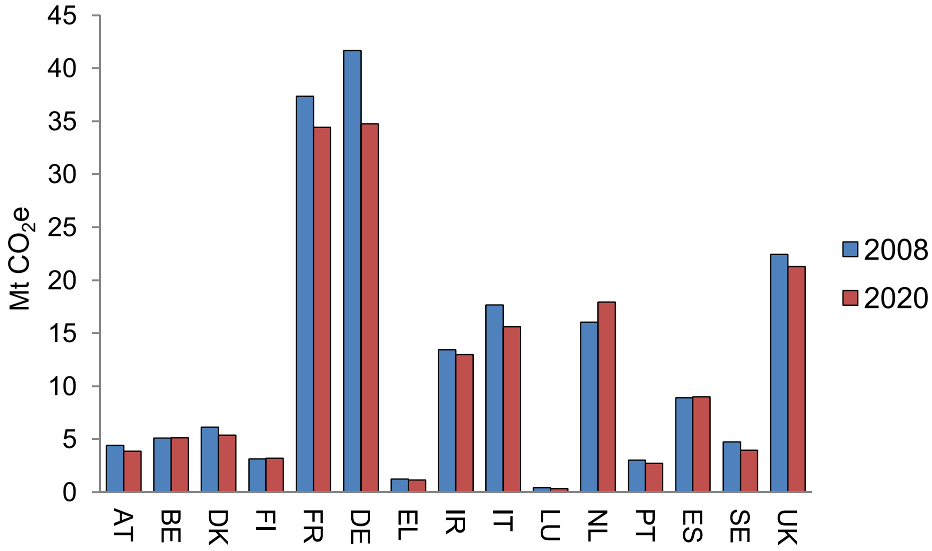

Figure 5.4: Total GHG emissions from the dairy sector for EU-15 member countries in 2008 and 2020

GHG emissions per unit product will decrease from 1.49 Kg CO_2e per L milk in 2008 to 1.30 Kg CO_2e per L milk in 2020 (Figure 5.5). The main dairy producers are the largest contributors to total GHG emissions and show below average unit emissions except for France and the United Kingdom in 2020 (Figure 5.5).

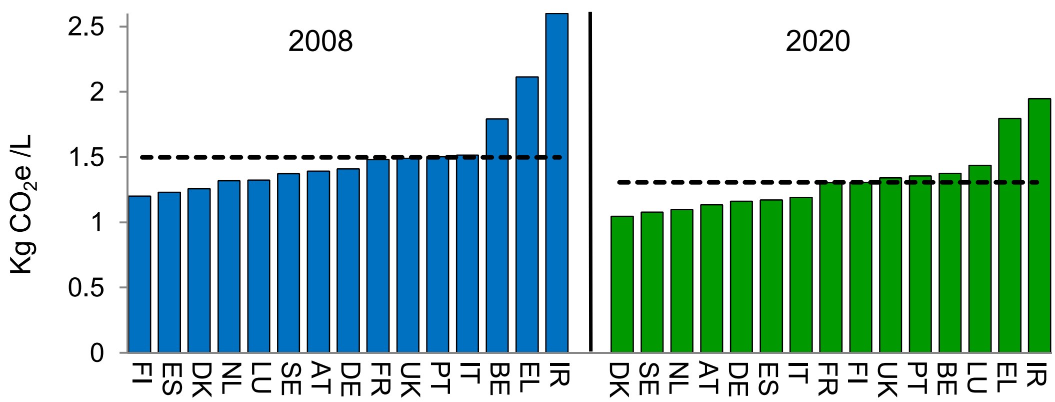

Figure 5.5: GHG emissions per litre milk production in 2008 and 2020. The dashed lines describe EU-15 averages in 2008 and 2020.

Per source contribution to total emissions is highly variable amongst member countries in 2020 (Figure 5.6). However, contribution from enteric fermentation and soil N_2O to total emissions are highest for all countries except Finland; indicating the importance of GHG reduction from these emissions sources. In 2020, SOC sequestration decreases GHG emissions by 3.5 Mt CO_2e in EU-15, but Austria, Belgium, Ireland, Luxembourg and Netherlands are soil C sources (Figure 5.6).

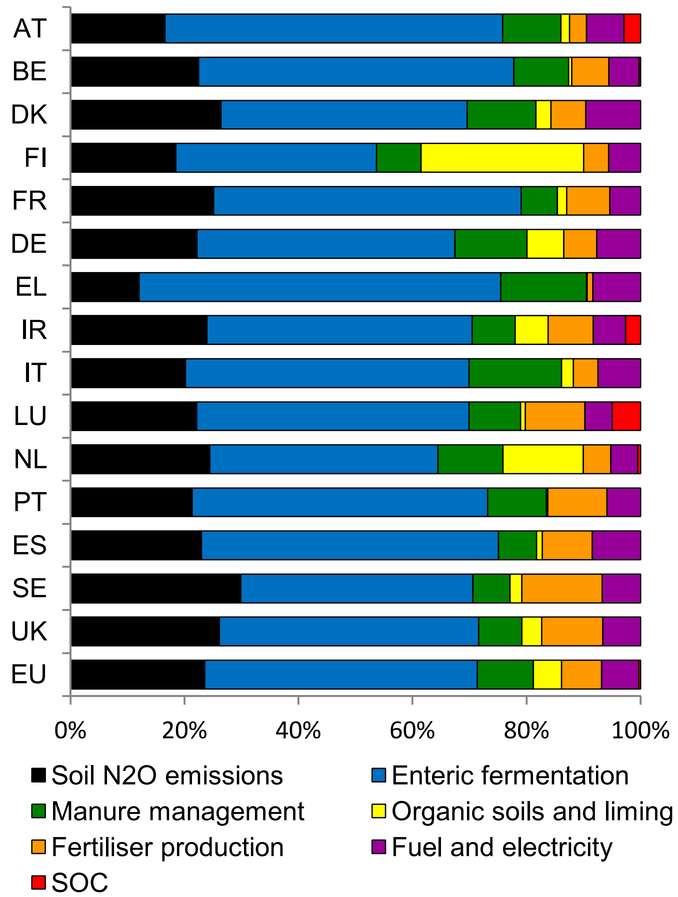

Figure 5.6: Relative contribution of GHG sources to total GHG emissions per country in 2020. Countries without SOC emissions are C sinks.

5.3.2 Technical reduction potential of mitigation measures

The total abatement potential of all mitigation options combined is 19.8 Mt CO_2e i.e. 12% of sector's GHG emissions in 2020. Most prominent mitigation options are M5, M6 and M7 with reduction potentials of 4.7 Mt CO_2e, 6.7 Mt CO_2e and 3.2 Mt CO_2e, respectively (Table 5.4). This also expresses the GHG emissions reduction per L milk i.e. 5%, 4% and 6%, respectively (Table 5.4). Although M6 shows the highest total GHG reduction, the per unit

GHG reduction is lower than to M5 and M7 as both of these measures increase milk yield; hence total GHG reduction potentials are less expressive than reduction potentials per unit of milk (Table 5.4).

Amongst agricultural measures, M2 is the most prominent with a reduction potential of 1.5 Mt CO_2e in 2020 (Table 5.4). M1 is only applicable in Finland, Portugal and Spain and shows together with M4, the lowest total GHG emission reduction across EU-15. For M1, lower dairy stock, increased N demand and N use efficiency of higher yielding crops will compensate for higher N fertiliser input in 2020 (Table 5.1 and Table 5.4). However, total emissions are reduced by around 4% compared to GHG baseline emissions in countries in which this mitigation option is applied. For M4, the total GHG reduction potential is low since NVZs cover large parts of European agricultural cropping area and thereby strongly decreasing the potential applicability of the measure.

Table 5.4: GHG reduction potential of mitigation options per member country and EU-15 totals. Empty cells indicate no GHG reduction by implementing mitigation measures.

Country	M1 - Optimal fertilis-ation	M2 - Reduced tillage	M3 - Cover crops	M4 - Restricted fertilisation during winter	M5 - Dietary lipids	M6 - Dietary nitrate	M7 - Dietary tannins	M8 - Dietary probiotics	M9 - Animal selection
AT*		-55		-6	-129	-184	-89	-30	-33
BE*		-124	-141	-14	-168	-241	-117	-39	-17
DK*		-38	-13	-14	-158	-225	-109	-36	-16
FI*	-117	-81	-42	-2	-67	-96	-47	-16	-7
FR*		-150	-47	-165	-906	-1294	-627	-209	-234
DE*		-683	-237	-60	-1093	-1561	-757	-252	-113
EL*		>-1		>-1	-38	-54	-26	-9	-10
IR*		-25	-7	-20	-309	-442	-214	-71	-80
IT*		-18		-40	-481	-688	-333	-111	-124
LU*		-10	-2	-1	-8	-12	-6	-2	-1
NL*		-157	-148	-56	-545	-778	-377	-126	-56
PT*	-104	-14	-9	-1	-80	-114	-55	-18	-21
ES*	-417	-24	>-1	-14	-183	-262	-127	-42	-47
SE*		-48	-7	-18	-97	-139	-67	-22	-10
UK*		-33	-41	-65	-426	-609	-295	-98	-44
EU-15*	-638	-1460	-729	-476	-4688	-6699	-3246	-1081	-813
EU-15†	1.30	1.29	1.30	1.30	1.23	1.25	1.22	1.27	1.28

Mitigation potentials are in kiloton (Kt) CO_2e (*) or Kg CO_2e / L milk (†).

5.4 Discussion and conclusion

5.4.1 Baseline development

In 2008, the dairy sector and its affiliated land use emitted 5% of total EU-15 GHG emissions (excluding soil C sink; EEA, 2010); attributed to the large contribution of dairy and beef products to total protein intake in Europe (Westhoek et al., 2011). Despite increasing production levels in 2020, carbon footprint per litre milk was reduced by 13%. Pelletier and Tyedmers (2010) claimed that a carbon footprint reduction of livestock activities by 13% per unit of livestock protein is required by 2050 to stay within sustainability boundaries. However, this study considered global GHG emissions for livestock production which can occur outside the accountancy boundaries of the present study. In 2020, the baseline soil N_2O emission reduction indicates that N surplus in European soils is projected to decrease (Bouwman et al., 2013). Further, dairy production efficiency per unit CO_2e increases without increasing land use for feed production. This indicates reduced negative impacts on the environment which is a promising development for the need to meet globally increasing demand for dairy products. However, further research should assess the impacts on the environment, rural populations, biodiversity and animal welfare in order to claim that the EU-15 dairy sector intensifies in a sustainable way (von Keyserlingk, 2013).

In line with increasing production levels and decreasing GHG emissions per unit of product, protein-rich feed, grass and straw compound of cow's diet (EU-15 average) is increasing by 16%, 17% and 20% between 2008 and 2020, respectively. Literature is inconsistent whether concentrate based diets reduce GHG emissions per unit of product (Bell et al., 2011) since high energy diets decrease enteric CH_4 emissions by increasing production per unit of energy intake (Gill et al., 2010) or increases GHG emissions if including emissions from LUC for soybean production (O'Brien et al., 2012). The differences might even be negligible (Kristensen et al., 2011). "Cradle to retail" studies state a GHG reduction per Kg of fat and protein corrected milk (FPCM) by increased cow's productivity; but emission reduction strongly declined if productivity exceeds 6000 kg FPCM per cow (Gerber et al., 2011). Zehetmeier et al. (2012) claimed that higher yielding cows have a proportionally higher human-edible plant protein intake. Although this is contrary to our findings i.e. proportion of feed cereals, fodder maize and protein-rich feed intake remain stable in 2020 as compared to 2008, it might be valid as our results can be attributed to shortcomings of feed intake projections. Human edible compounds in cow's diet are an issue of concern and reducing their share by replacement and increased feed utilisation efficiency should be prioritised since human edible plants show the highest efficiency in calorie transmission to humans (Gill et al., 2010). Land use for feed production also may become a competitor for human food production, besides bioenergy crop production and nature conservation (Herrero and Thornton, 2013; Smith et al., 2010).

Livestock intensification beyond a certain threshold may thus not deliver the solution for reducing the dairy sector's environmental impact per se, also in terms of consumers' environmental and ethical concerns about high intensive livestock production and food waste through over production (Bellarby et al., 2013).

5.4.2 Mitigation options

M1 and M4 reduce production inputs and decrease environmental pollution while maintaining production levels and therefore these mitigation options should be considered despite their relative low GHG reduction potential. However, MITERRA-Europe assumes a loss of 40% of total N input to the soil through atmospheric deposition, leaching and runoff and thereby NH_3 emissions and ground water pollution is higher than indicated by the application potential of these measures and is this is still an issue of concern.. Robertson and Vitousek (2009) claimed that a wrong timing of fertilisation that mismatches with the crop N need is probably the most significant reason for N loss through excess application in annual cropping systems. In this regard the global fertiliser industry developed and supports fertiliser best management practices through the 4R Nutrient Stewardship initiative i.e. setting guidelines for the right nutrient source, at the right rate, right time and the rate place (Snyder et al., 2014). Combining the global position system and the geographic information system with advanced fertiliser application technologies and decision support systems can optimise the rate and timing of fertiliser application and thereby reduce N excess (Snyder et al., 2014).

M2 showed only a small C sink potential in cropland since the potential application area is small. However, Powlson et al. (2014) claimed a general over-estimation of C sequestration potentials for no or reduced tillage practices and the additional small GHG reduction through reduced fuel combustion might be offsets by a possible increase in soil N_2O emissions. In this study no effect on N_2O emission were assumed due to inconsistent study results. As discussed in section 2.3.2.2 reduced tillage can increase soil N_2O emissions if replacing conventional tillage in dry climates. If additionally considering a potential crop yield decrease, N_2O emission per unit of crop yield can increase in short term significantly but in long-term (≥ 10 years) it was shown that yield-scaled N_2O emissions were lower compared to conventional practices systems (Kessel et al., 2013). To allow for the full abatement potential from this mitigation option, it must be operational over long period. However, soil N_2O emissions could be reduced by applying fertiliser in a depth of > 5cm to the soil instead of surface application (Kessel et al., 2013). Another issue of concern is that unlike CH_4 and N_2O emission reduction from livestock or soil, increasing SOC storage is not permanent, and this is a critical issue that also requires consideration (Lal, 2004).Enteric CH_4 is the largest GHG source in 2020 and should be considered in the mitigation mix (Figure 5.6). Focus on enteric CH_4 reduction can increase production efficiency per DMI (van Middelaar et al., 2014), increase yield by application of M5, M7, M8 and M9 and show promising reduction potentials (Figure 5.5). In terms of intensification, feed supplements could replace human edible feed compounds or concentrate compounds in the diet and M9 could replace selection for higher cow yield as this leads to declining fitness traits i.e. fertility and health (Bell et al., 2011); hence these measures should be considered for intensification. Zehetmeier et al. (2012) warns of milk yield increase since it reduces total dairy stock to maintain milk production levels and thereby reducing beef production from the dairy sector and increasing GHG emissions from beef farming since demand needs to be satisfied. However, there are several issues with mitigation through feed supplementation. Here these are dsicussed for the dietary mitigation options that decreased CO_2e per output most strongly i.e. dietary lipids and tannins. We assumed an increased milk production of 5% by lipid supplementation but this is a simplification and this may not be the case for all cows in EU-15. The meta-analysis of Eugene et al. (2008) containing 7 publications could not show an effect on milk yield. However, milk yield response could be dependent on the main forage in the diet as Grainger and Beauchemin (2011) stated that maize silage based diets did not show increasing milk yield, contrary to alfalfa based diets. Weiss and Pinos-Rodriguez (2009) showed in their experiment by adding saturated fatty acids to 72 Holstein cows that high forage diets (67% corn silage and 33% alfalfa silage) did not change milk yield and with low forage diets milk yield increased by 2.6kg per day. Milk responses to fat supplementation are complex depending amongst others on the type of lipid (e.g. saturated or unsaturated) and the basal diet of the cow (Grainger and Beauchemin, 2011). Inconsistencies of dietary lipids in CH_4 reduction have been discussed in section 2.3.1.1. Further, there is the issue of persistency of CH_4 reduction by lipid supplementation (Woodward et al., 2006) but this requires further research. Lipid supplementation is limited for pasture based systems. However, grasses could be used that contain high levels of fat e.g. in New Zealand a transgenic approach is used to increase fat content in leaves of rye grass by up to 40%; alternatively fats could be added to the drinking water of the cows (Grainger and Beauchemin, 2011).

For M7 there is a high variability of CH_4 reduction (as discussed in section 2.3.14.). Based on meta-analyses we assume a moderate level of GHG reduction but it needs to be considered that different tannin agents show varying GHG reduction potentials and hence not all types of tannins are recommended for application. Future research must understand the effect on the N content in manure after adding tannins to the diet as reduced N content could compromise crop yield if the manure is applied to the field (Montes, 2013). Tannin application at an overdose can be toxic to the animal but this is effect was not observed for some hydrolysable tannins which show also stronger CH_4 reduction potential compared to CT (Goel and Makkar, 2014). In this study extracted tannins from chestnut for feed supplementation were considered. For pasture based systems, tanniferous forages can be introduced to the pasture. The meta-analysis of Archimède et al. (2011) containing 22 *in vivo* studies on tanniferous forages showed that C3 grasses produced up to 17% less CH_4 per organic matter intake compared to C4 grasses and are thus as effective as legumes from cold climates in reducing GHG emissions. Legumes from warm environments showed higher CH_4 reduction than cold legumes. Based on these results tannin rich plant species should be selected for pasture introduction. Additional benefits of tanniferous forages can be improvement of silage quality, ruminant productivity and health (Gerber et al., 2013a). Generally introduction of tanniferous forages poses the advantage that tannins do not need to be extracted which can be a costly process. Therefore, this should be considered as an alternative mitigation option.

GHG emissions related to production and transportation of feed supplements were not considered, but these factors could affect GHG reduction potentials (Williams et al., 2013); especially in case of LUC for lipid production. Further, simultaneous implementation of mitigation options at farm scale can also reduce total abatement potentials, and this needs to be considered by decision processes. Attention should be given to considerably varying abatement potentials of mitigation options across strongly heterogeneous dairy farm systems within EU-15 as already indicated by strong variation in abatement potentials across EU-15 member countries. Therefore, measure implementation should be site specific while considering characteristics of individual dairy farms.

5.4.3 Limitations

GHG emission estimates from enteric fermentation and manure storage systems are high in comparison to the European GHG inventory for 2008 (EEA, 2010) that stated 44 Mt CO_2e and 8.4 Mt CO_2e from these sources, respectively. This is mainly explained by inclusion of GHG emissions from all dairy cattle and their heifers without correcting for beef production. Thereby, the estimated 1.49 Kg CO_2e per litre milk in 2008 are higher as compared to studies that only considered dairy production e.g. Weiske et al. (2006) and Lesschen et al. (2011) which stated 1.4 and 1.3 Kg CO_2e per litre milk in European dairy production, respectively. With respect to the research objective in this chapter, this is a valid approach since the focus was on baseline GHG emissions development and technical abatement potential of mitigation options of the total dairy sector. However, this needs consideration when interpreting the results.

In the following main uncertainties are identified that are unique to this assessment. First, MITERRA-Europe includes inputs from various sources that may cause issues with comparability and harmonisation of data as this can be seen compared to the European GHG inventory that is based on national statistics. For instance, the European GHG inventory reported dairy cow stock of 19.5 M heads (EEA, 2010), while MITERRA-Europe stated 18 M heads in 2008. Further, at national level correcting for import and export of cows between EU member countries is subject to major uncertainties and might also lead to systematic errors (Lesschen et al., 2011). Second, data gaps in MITERRA-Europe had to be filled with assumptions. Most critical assumptions are: i) area of the three different grassland types and their yields; ii) estimation methodology for N-leaching fraction; iii) baseline adoption levels and geographical distribution of mitigation options and iv) future activities of the dairy sector. Finally, MITERRA-Europe relies on a Tier 1 approach to estimate GHG emissions from several sources e.g. manure storage system, SOC change and GHG reduction potentials of animal mitigation options. Oversimplification by fixed EFs and reduction potentials at high spatial level does not account for spatial or temporal variation and may lead to miscalculations (Crosson et al., 2011). Kros et al. (2012) did a MC simulation of N emission and N runoff for estimates from the INTEGRATOR model that is similar to MITERRA-Europe. This study revealed that N_2O emissions show output uncertainty of 12% on EU-27 level with strong variations within some countries. Such an approach would be a valuable contribution to inform about uncertainty and to guide our modelling approach to more robustness.

5.5 Conclusion

An innovative approach was developed that integrates information from various data sources and different economic models to generate BAU activity levels in 2008 and projections until 2020 for the dairy sector in the EU-15. These data were combined in the MITERRA-Europe model to estimate baseline GHG emissions and technical reduction potentials for nine mitigation options in 2020.

In 2008, the EU-15 dairy production contributed to 5% of the total EU-15 GHG emissions. However, total GHG emissions and carbon intensity of production is estimated to decrease by 8% and 13% in 2020, respectively. Despite this promising development, further GHG reductions are possible through the proposed mitigation options by 2020. Optimal fertilisation (M1) and reduced fertilisation during winter (M4) showed to be suitable for sustainable intensification as GHG emissions and simultaneously production inputs are reduced. However, mitigation should focus on GHG emissions from enteric fermentation as this source contributes to 48% of the total emissions in 2020. Therefore, the possible contribution to sustainable intensification of feed addition of lipids (M5), nitrate (M6) and tannins (M7) were highlighted as they facilitate large GHG reduction potentials, increased production efficiency (for M5 and M7) and potential replacement of unfavourable feed compounds in the diet.

Chapter 6 - Assessing the economic mitigation potential for greenhouse gas emissions and its uncertainties in the European dairy sector

6.1 Introduction

Despite of many MACCs having been developed for agricultural sectors in different European member countries, there is still a lack of European-wide MACCs focusing only on the dairy sector while using an unified methodology for each member country. This would allow direct comparison of the abatement potentials of EU member countries and enable the European Commission to formulate legislation on GHG reduction targets, agricultural policies or the European Trading Scheme (ETS) for the European dairy sector. As discussed before it is further important to assess uncertainties that occur during the MACC exercise. A popular approach to tackle parameter uncertainty in models has been to undertake traditional spreadsheet based sensitivity analysis. Nevertheless, when a model involves more than a few uncertain parameters, a spreadsheet based sensitivity analysis can become unmanageable and cumbersome. Therefore, a more feasible and rigorous alternative method for tackling parameter uncertainty was used i.e. MC simulation (Vose, 2000).

This chapter aims to estimate the abatement potential for 9 mitigation options and shows the information sources utilised to estimate the economic abatement potential in a transparent manner. Based on the MACC results, a MC simulation was undertaken to assess model input and output uncertainty. The first section describes the transformation of findings from chapter 5 for the purpose of a MACC development and the methodologies for the MC simulation. Thereafter, the results of the MACC exercise and MC simulation are shown. The final section discusses the findings for the mitigation options in comparison to other studies and evaluates the importance of the uncertainty assessment.

6.2 Methodologies

6.2.1 Adjusting the EU-15 dairy sector baseline

This MACC exercise considered baseline activities from 2009 to 2020 and associated GHG emissions. The baseline scenario developed in chapter 5 was extended by additional economic and statistical data sources to allow for estimation of measure implementation costs. Based on the annually published dairy reports (DG-AGRI, 2011 - 2014), the databases of the International Farm Comparison Network (IFCN; Hemme, 2012) and Farm Accountancy Data Network (FADN), the number of dairy farm holdings were set and economic profiles for each country specific model farm were generated. These average model farms only represented specialist's dairy farms (with >50% revenue from dairy production) adding up to around 80% of total EU-15 dairy production (varying per country). Since baseline GHG emissions for total EU-15 dairy production including land area for feed production were estimated in chapter 5, these estimates had to be adjusted to account only for specialist's dairy farms with their on-farm food production. This led to a GHG emission reduction from animals by roughly 10% and from land use by 60% in EU-15 average (Table A.1). The CAPRI forecasts were expanded by the DG-AGRI medium-term baseline (DG-AGRI, 2012) e.g. for milk yield developments and historical developments from Eurostat between 2001 and 2012 (EC, 2014b) e.g. for dairy farm holding number projections (Table A.1).

The price projections in 2020 for consumer and producer goods were based on several sources to facilitate robustness for this exercise: i) historical commodity price development published by Eurostat (EC, 2014b), European Commission (EC, 2014c), FADN and indexmundi (http://www.indexmundi.com/commodities) and the commodity price forecast of OECD/FAO (OECD-FAO, 2009 - 2014) and of the World Bank (World-Bank, 2015). Price forecasts were prioritised based on historical development that were country specific and available for at least the last ten years over international forecasts from the OECD/FAO and the World Bank (Table A.2). These price projections were adjusted by future inflation rate to allow comparison to 2009 prices.

6.2.2 Assessing GHG reduction potentials and costs of measure implementation

Similar to the baseline estimation, mitigation potentials for each mitigation option as assessed in chapter 5 were adjusted. Application of a mitigation measure may change input costs (feed purchase, fertilizer application, plant protection, seeds, electricity and fuel), investment, labour, machinery, energy/fuel input and/or yield (Table 6.1). The implementation costs were estimated specifically for each country; hence CE for single mitigation options varied as compared to the EU -15 averages. The implications of implementing a measure on-farm were based on scientific literature reporting these for dairy farms, expert judgment and the MITERRA-Europe model, which estimated the area of application and changes in fertilizer application and fuel consumption. A social discount rate of 3.5% was used to simulate the NPV of the measure's lifetime costs in the benchmark year 2009.

Modelling possible interactions between simultaneously implemented mitigation options is a very complex process for a large region like Europe. Interaction between following mitigation options were expected for i) optimal N fertilisation (M1) and reduced organic fertilisation during the wet season (M4); ii) reduced tillage (M2) and cover crops (M3) and iii) all animal feeding strategies (M5 – M8). To avoid potential interaction between the measures in each set, the adoption potential of the measure did not overlap: i) farmers did not apply inorganic fertiliser during the winter season without cash crops being cultivated and organic fertilisation during winter time is only prohibited without crops being grown; ii) although both measures are applied during the winter season, these measures are not applied on the same cropping area; iii) it was assumed that each cow receives only one feed additive i.e. the sum of the maximum technical adoption of all these measures does not exceed 100% of the cows.

Table 6.1: Cost implication for the mitigation options. Negative costs refer to cost savings.

No	Cost factors		
	Investment	Yield	Main input and output factors (in 2020) †
M1	Management of optimal fertilisation rate	Crops: -2%/ha	Fertiliser: -69€/ha* Feeding: 23€/head Management 2.7€/ha
M2	Power harrow for reduced ploughing depth		Plant protection: 3€/ha Fuel consumption: -21€/ha Machinery use: -7€/ha New equipment: 1890€/farm
M3	Cover crop seeds, fuel		Plant protection: -6€/ha Fuel consumption: 17€/ha Machinery use: 6.4€/ha Labour: 4€/ha New equipment: 22€/ha
M4	Manure storage for storing additional manure during winter		Fertiliser: -8€/ha Fuel: -10€/ha Machinery use: - 7€/ha Labour: - 4€/ha New equipment: 10€/head
M5	Lipids	Milk: +5%/head Body weight: +2%/head	Milk production: -126€/head Cow culling: -10€/head Feeding: -19€/head Additive: 170€/head
M6	Nitrate		Feeding: -12€/head Additive: 43€/ha
M7	Tannins	Milk: +5%/head Body weight: +5%/head	Milk production: -125€/head Cow culling: -22€/head Additive: 345€/head
M8	Probiotics	Milk: +3%/head Body weight: +3%/head	Milk production: -75€/head Cow culling: -13€/head Feeding: 17€head Additive: 69€/head
M9	Management of breeding programme	Milk: +10%/head	Milk production: -251€/head Feeding: 29€/head Management: 47€/head

* Fertiliser reduction only in Finland, Portugal and Spain. The costs represent the average costs in these three countries.
† These factors are European average; therefore cost factor in each member state may vary.

6.2.3 Activity levels and adoption potential of mitigation options

In this chapter, the uptake potentials by farmers were defined and added to the adjusted BAU activity levels of the mitigation options as assessed in chapter 5. This is a more subjective judgment compared to assessing the BAU activity levels. Within the project 'Policy Incentives for climate change mitigation techniques project' of the the sixth framework programme, the report of Lesschen et al. (2007) published potential degree of cropland measure implementation (M1 – M4) in Europe. These estimates were utilised for this study. For the livestock measures (M5 – M9), there are no adoption potential estimates available. Therefore, assumptions were based on expert judgement and literature (Beauchemin et al., 2008; Franz et al., 2010). In general, it was assumed that farmers are less likely to implement very costly mitigation options, those that imply risks for the yield by applying an overdose i.e. M5 and M6 and those that show large adoption during the BAU scenario. The highest adoption rate is therefore expected for M8, followed by M6 and M7 and lowest for by M5 and M9 since application is high during the BAU scenario (Table 6.2).

The adoption rate shows for most measures a sigmoid curve as studies in the agriculture sector have shown that adoption starts slowly, increases strongly until the adoption rate declines strongly (Läpple and Rensburg, 2011; Llewellyn et al., 2012). For this MACC exercise, the adoption scenarios as described by Pellerin et al. (2013) were adopted. This resulted in sigmoid adoption curves for M1, M2, M3, M5, M6, M7 and M8. For M3 and M9, a linear adoption was assumed since these measures are already widely applied, and there was no need to account for early adoption rates. A similar shape for sigmoid curves as elaborated by Pellerin et al. (2013) were generated but adjusted since the simulation period in this MACC exercise is only 11 years (Table 6.2). For M6, the adoption started in 2015 since it is assumed that nitrate feeding supplements became available in Europe since this year. An adoption curve for M7, M8 and M9 is not included in the study of Pellerin et al. (2013); for M7 and M8 a similar adoption rate as for M5 were assumed since they are also feed additives with low BAU activity levels.

Table 6.2: Adoption potential of the mitigation options.

No.	BAU adoption (in 2009)	Possible additional adoption (in 2020)		Adoption scenario†
		Area or head	Share of total (%)	
M1	11.71 M ha*	1.03 M ha	7.9	Sigmoid adoption: maximum technical adoption in 2020; sigmoid curve: 10% adoption after 3 years; 90% adoption after 8 years.‡
M2	0.57 M ha	0.74 M ha	34.8	Sigmoid adoption: maximum technical adoption in 2020; sigmoid curve: 10% adoptin after 2.5 years; 90% adoption after 8.5 years.‡
M3	0.59 M ha	1.33 M ha	62.8	Sigmoid adoption: maximum technical adoption in 2020; sigmoid curve: 10% adoptin after 2.5 years; 90% adoption after 8.5 years.‡
M4	4.35 M ha	6.99 M ha	53.4	Linear adoption‡
M5	4.37 M hd	3.06 M hd	21	Sigmoid adoption; sigmoid curve: 10% adoption after 2 years; 90% adoption after 8 years.‡
M6	0	4.37 M hd	30	Sigmoid adoption; sigmoid curve: 10% adoption after 6 years; 90% adoption after 9 years.‡
M7	0.73 M hd	4.15 M hd	28.5	Sigmoid adoption; sigmoid curve: 10% adoption after 2 years; 90% adoption after 8 years.‡
M8	2.91 M hd	6.99 M hd	48	Sigmoid adoption; sigmoid curve: 10% adoption after 2 years; 90% adoption after 8 years.‡
M9	12.13 M hd	2.43 M hd	16.7	Linear adoption‡

* No adoption possible in following countries: Austria, Belgium, Denmark, France, Germany, Greece, Ireland, Italy, Luxembourg, Netherlands, Sweden, United Kingdom.
† The adoption scenarios where adoption from Pellerin et al. (2013). The sigmoid curves have the same shape but in this study the Maximum technical applicability is reached after 11 years.
‡ Maximum technical adoption reached in 2020.
Sources: (Pellerin et al. (2013), van Arendonk and Bijma (2003), Velthof et al. (2009) and Eurostat.

6.2.4 Uncertainty assessment

The robustness of measure's CE and abatement potential at negative costs were examined by taking into account uncertainty of parameter inputs. The MC simulation via Vose ModelRisk Professional v4.3 included 10000 runs. The sensitivity of all input parameters were assessed, except the technical abatement potential of the mitigation options, the GWPs of different GHGs and the baseline activity levels of mitigation options. This is due to the fact that these were used exclusively by the MITERRA-Europe model and could not be included in the MC simulation. For the uncertainty assessment, a consistent pdf for all input parameters were chosen i.e. triangular distribution and a pdf range of +/- 15% of the original parameter value since some model input variable did not allow for a precise pdf generation and it would effect the results of the MC negatively if some variable are treated differently to others. The pdf range may vary if this range was not viable e.g. the cost for additional labour input becomes a value close to zero or negative.

For ranking the uncertainty contribution of the input parameters to the total uncertainty of the abatement potential at negative costs, all input parameters were seperated in 3 key input categories: baseline development, cost of mitigation options and adoption potential for mitigation options (Table 6.3). These input categories refer to certain data types: i) statistical i.e. historical data of prices and activity levels; ii) economic i.e. forecasted dairy sector's activity and economic impact of measure implementation; iii) behavioural i.e. farmer's choices for mitigation options which drive the adoption potential (Table 6.3). The methodology for the ranking was adopted from Eory et al. (2014). First, the uncertainty of the abatement potential at negative costs for all input parameter and each input category were estimated individually. Thereafter, the 95% confidence interval (CI) of each of the four results were identified and divided by the mean abatement potential.

Table 6.3: Description of input parameter categories and type of data included in this MACC.

Input category	Description	Type of data
Baseline development		
Farmers activity level	Current activities of dairy farmers in 2009	Statistical
Farmers activity forecast	Modelled activities of dairy farmers until 2020	Economic
Prices	In- and output prices	Statistical
Price forecast	Forecast of in- and output prices	Economic
Costs of mitigation options	Impact of the mitigation options on the farm's economic performance	Economic
Discount rate	Choice of discount rate to estimate NPV	Economic
Adoption potential	Uptake of mitigation options by farmers that is additional to BAU activities	Behavioural
Forecasted changes	Behavioural change of Farmer's willingness to implement mitigation options	Behavioural

6.3 Results

6.3.1 GHG emissions in the baseline scenario

Figure 6.1 shows that total GHG emissions from the EU-15 dairy sector (accounting for specialist dairy farms) are declining by 9% (10.9 Mt CO_2e) between 2009 and 2020; mainly due to decreasing cow number of around 10% by 2020. N_2O emissions from soil are 17.5 Mt CO_2e in 2020 with a decline of 1 Mt CO_2e compared to 2009 levels. Enteric CH_4 emissions are 75 Mt CO_2e in 2020 with a reduction of 8.8 Mt CO_2e as compared to 2009 levels, emissions from manure are 15.3 Mt CO_2e in 2020 with a decrease of 1.1 Mt CO_2e as compared to 2009 levels. Soil C sequestration leads to a reduction of 0.4 Mt CO_2e in 2020.

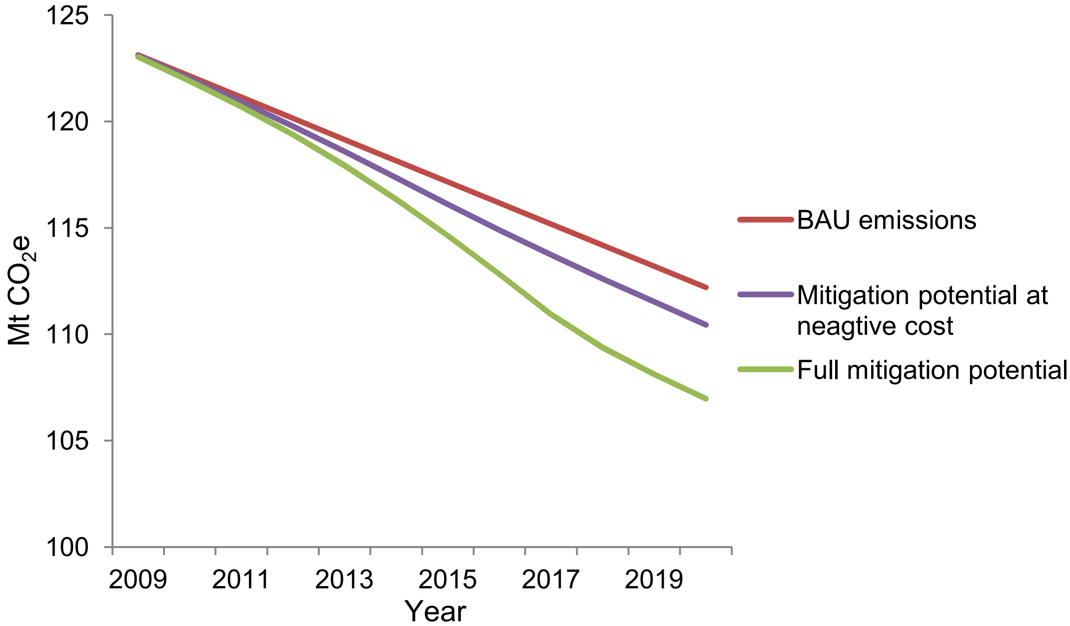

Figure 6.1: GHG emission development for the BAU and abatement scenarios. Three mitigation scenarios are illustrated: GHG reduction of the mitigation options with negative CE, GHG reduction of all mitigation measures and mitigation potential at costs of 15€/tCO_2e.

6.3.2 Mitigation potential and cost-effectiveness of abatement in 2020

The measure's abatement rate, mitigation potential, implementation cost and CE of abatement for the year 2020 are shown in Table 6.4. The abatement rate for the measures M1-M4 ranges between 0.02 – 0.09 tCO_2e/ha in 2020. Optimal fertilisation (M1) shows the highest abatement rate with 0.09 tCO_2e/ha while reduced fertilisation during the winter season (M4) shows the lowest abatement rate 0.02 tCO_2e**.** The abatement rates of the livestock measures are 0.1 – 0.42 tCO_2e/head in 2020. The highest abatement rate shows dietary lipids (M5) and dietary nitrate (M6) with 0.42 tCO_2e per head in 2020 due to the significant enteric CH_4 reduction potential.

The maximum adoption is highest for reduced tillage (M2), and lowest for M1 since large crop areas in Europe are under conventional tillage and balanced fertilisation is only implemented in Finland, Portugal and Spain (Table 6.2). For the livestock measures dietary probiotics (M8) show the highest adoption potential, and animal selection (M9) shows the lowest adoption potential; according to a low implementation barrier for introducing probiotic feeding amongst farmers and animal selection being a common practice in the dairy sector (Table 6.2).

The measures M1, M2, M4, M8 and M9 show negative CE or i.e. cost savings for the farmers (Table 6.4). M4 shows the lowest CE with -475.8€ per t CO_2e abated and a total mitigation potential of 100 Kt CO_2e in 2020. Dietary nitrate (M6) shows the highest abatement potential of 1.5 Mt CO_2e at medium costs of 44.5€ per tCO_2e abated. M2 and M9 are the most prominent mitigation options with a mitigation potential of 0.4 Mt CO_2e and 0.7 Mt CO_2e in 2020, respectively; as they reduce GHG emissions at negative costs of -63.4 and -339.8 €/tCO_2e. In terms of costs for subsidies for farmers' compensation for implementation costs, implementation of M2 and M9 would generate an additional income for the farmers, hence no compensation would be required. M6 would require additional costs of 49.8 M€/% GHG reduction of the dairy sectors totals (Table 6.4). The mitigation options adding tannins (M7) and adding lipids (M5) have a high CE of 513.5 €/tCO_2e and 69.9 €/tCO_2e, respectively. Therefore, these mitigation options should not be considered for large-scale implementation.

Table 6.4: Average abatement rate, cost of abatement, adoption potential, cost-effectiveness and mitigation potential in 2020.

No.	Abatement rate*		Cost*		Cost effectiveness*		Mitigation potential*	
	tCO_2e/ha	tCO_2e/hd	€/ha	€/hd	€/tCO_2e	M €/% reduction†‡	Mt CO_2e	% reduction
M1	-0.09†		-22.2		-251.2	-281.8	0.1	0.08
M2	-0.08†		-4.8		-63.4	-71.1	0.4	0.32
M3	-0.08†		13.7		151.6	170.1	0.2	0.16
M4	-0.02†		-9.3		-475.8	-533.8	0.1	0.08
M5		-0.42		29.05	69.9	78.4	1.1	0.94
M6		-0.42		18.5	44.5	49.8	1.5	1.35
M7		-0.22		115	513.5	576.1	0.7	0.65
M8		-0.1		-1.4	-15	-16.9	0.5	0.44
M9		-0.3		-100.2	-339.8	-381.2	0.7	0.65

* refers to the year 2020.
† Abatement rate may vary in different crop- and grasslands.
‡ Note that these costs only occur if one % of total sectors GHG emissions would be reduced. Not all mitigation options do allow a reduction of one % and therefore the implementation costs of these mitigation options would be smaller.
±% reduction refers to abatement of one % compared to the total GHG emissions of the EU-15 dairy sector in 2020.

6.3.3 Abatement potentials in the EU-15 dairy sector

The full mitigation potential of the 9 mitigation options in 2020 is 5.3 Mt CO_2e i.e. 4.67% of total GHG emissions from the EU-15 dairy sector (Figure 6.1 and Table 6.4). There is a significant mitigation potential that can be achieved with increased income for the farmers i.e. 1.8 Mt CO_2e reduction (1.6% compared to total GHG emissions) in 2020 (Figure 6.2 and Table 6.4). According to this win-win scenario, the additional income for the farmers would be 343.4M € (discounted to 2009 prices) in 2020.

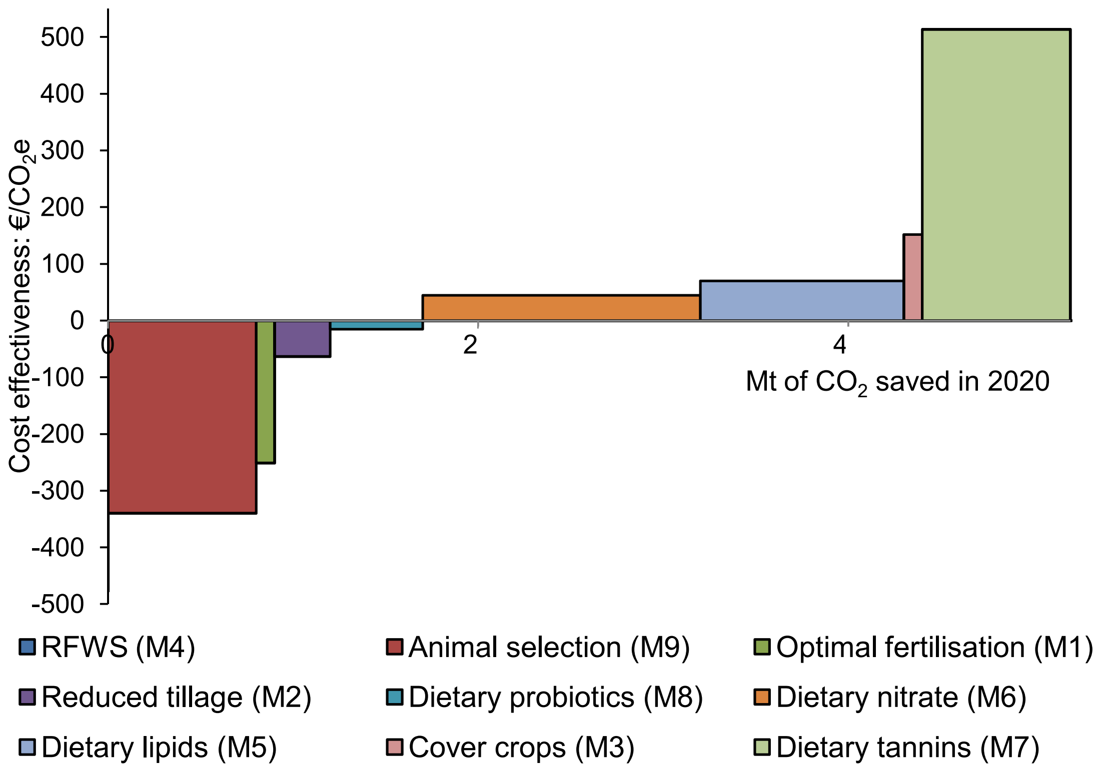

Figure 6.2: MACC for the EU-15 dairy sector in 2020. The Maximum abatement potential and CE are for the year 2020. The discount rate is 5%. This MACC is an EU-15 average.

Figure 6.3 shows the full abatement potential and the abatement potential at negative costs, specific for each EU-15 member country in 2020. The mitigation potential is highest for Germany, France and the Netherlands. These countries hold not only major dairy production systems but also have large cropping and grassland areas integrated into these dairy production systems. Greece, Luxembourg and Portugal show the lowest mitigation; since they have the smallest dairy sectors as compared to the other member countries. The share of abatement potential at negative costs relative to total abatement potential varies considerably within member countries. Finland, Germany, France, Sweden and Austria show the highest share of abatement at negative costs. Hence mitigation policies should target these countries.

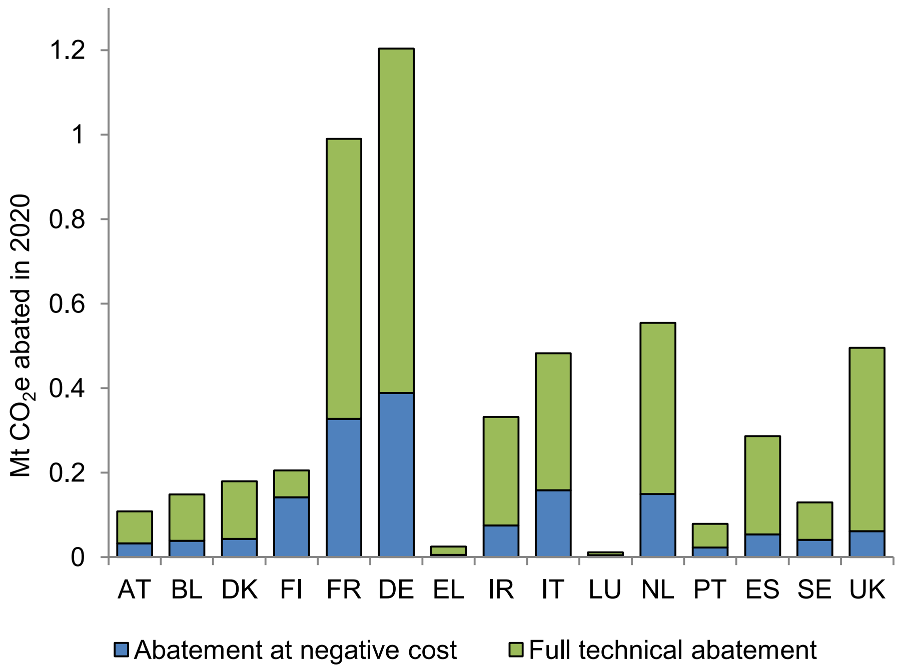

Figure 6.3: Full abatement potential and abatement potential at negative costs for each EU-15 member country in 2020.

6.3.4 Model input uncertainty

The uncertainty ranking of the input categories shows that ‘adoption potential’ is the most important and ‘baseline development’ the least important contributor to the total uncertainty of the abatement potential at negative costs (Figure 6.4). The category ‘adoption potential’ only includes behavioural data.

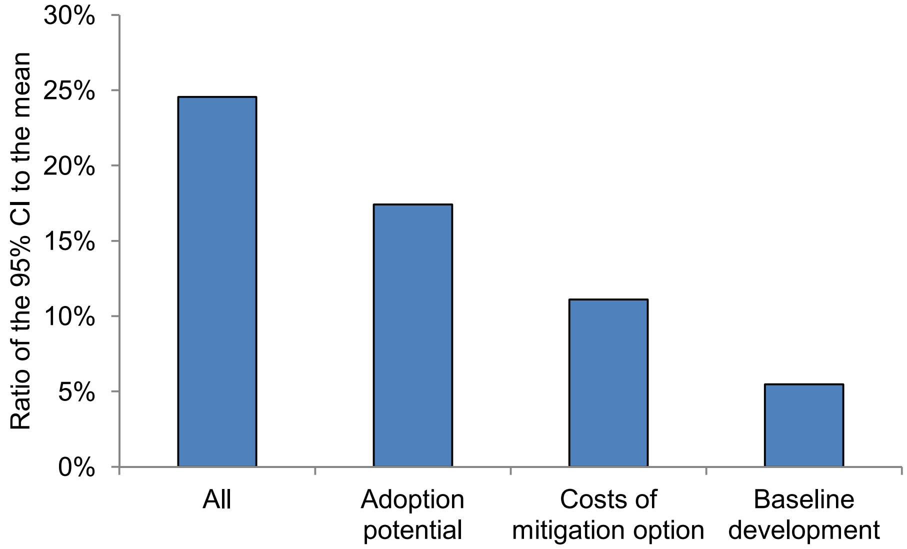

Figure 6.4: The uncertainty of individual input category and all input categories combined in a ratio of 95% CI to the mean of abatement at cost-negative abatement.

6.3.5 Uncertainties of the measure's CE

Figure 6.5 shows the uncertainty in the CE of measures. The uncertainty is high for M1, M5, M8 and M7. The uncertainty for M2 and M6 is low. The MC simulation reveals that only M9 and M4 are stable on the cost negative side and dietary tannins are stable on the cost positive side, while the CE of other win-win mitigation options may cross the cost-effectiveness of 0.

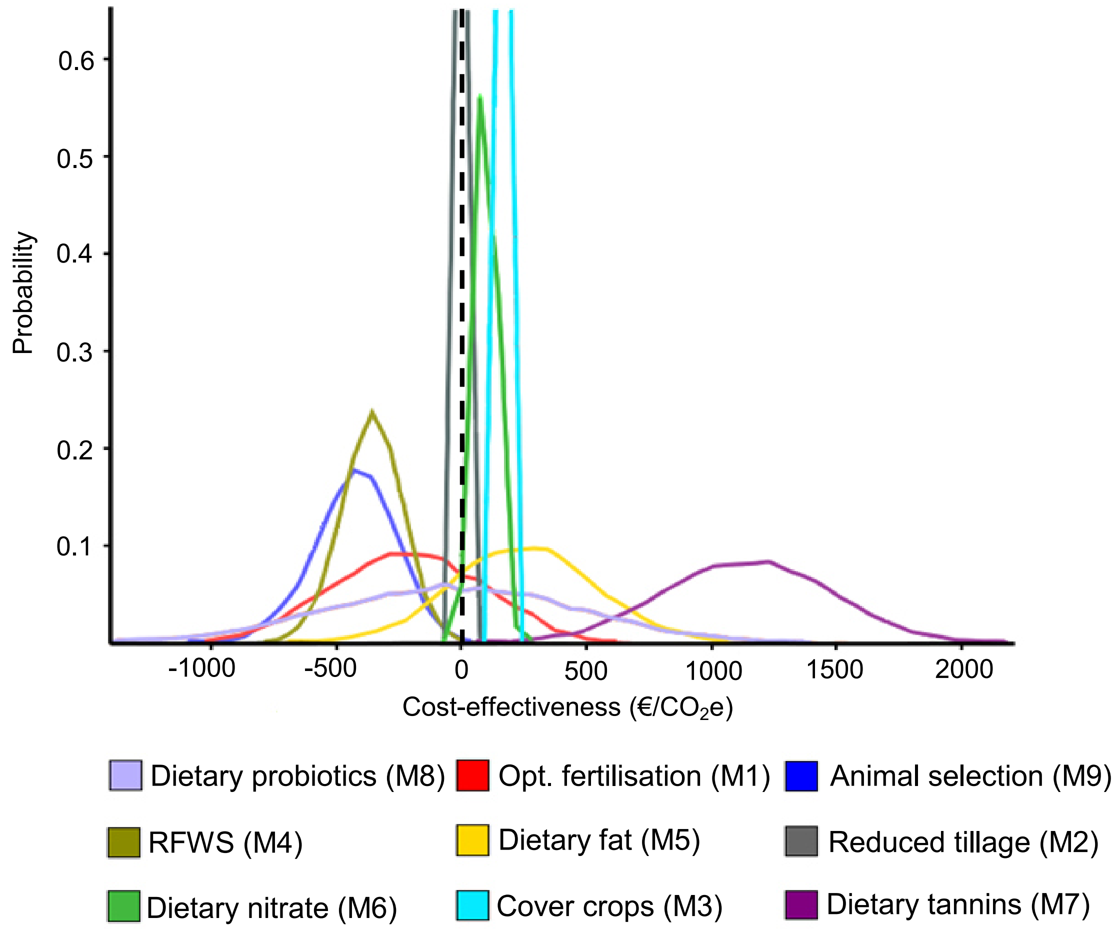

Figure 6.5: Uncertainty of the mitigation option's cost-effectiveness of abatement. The dashed line indicates a cost-effectiveness of 0. The abatement on the left side is cost negative and on the right side cost positive. The probability curve for reduced tillage and cover crops shows a sharp inclination until a probability of 1 and 0.8, respectively.

6.4 Discussion

The baseline GHG emissions decline by 9% between 2009 and 2020. This is a positive development. However, GHG emissions outside the farm gate were not accounted for. Based on a LCA, Leip et al. (2010) estimated that 21 % and 29% of total GHG emissions from European livestock production result from the energy sector and land use and LUC from mainly non-European countries, respectively. Considering the increasing concentrate feed share for high productive cows and these being largely imported from non-European countries, total baseline GHG emissions might increase if considering life cycle GHG emissions. Regarding energy related GHG emission for electricity generation and transport, these might not increase in future as energy efficiency is expected to increase strongly but this depends if the increasing production levels will outpace the improvements in energy-efficiency. This study showed significant mitigation potential in the EU-15 dairy sector that is achievable in an economically viable way. Policy makers tend to ignore the mitigation potential in livestock production. Substantial mitigation in this sector is required to adhere to the stringent GHG reduction targets of the EU. It is projected that EU-15 milk production level will increase continuously and with a significant increasing global milk demand between 2015 and 2030 i.e. 40% in developing countries and 5% in developed countries, it remains important for Europe as a key exporter of dairy products to reduce the carbon footprint of dairy production while simultaneously increasing its productivity (Thornton, 2010). It was shown that implementation of mitigation options M1, M2, M4, M8 and M9 are an adequate solution for this issue. However, the most immediate action by policy makers should be to enhance knowledge dissemination that allows farmers and consumers to be a driver of implementing these mitigation options in dairy production (Gill et al., 2010).

6.4.1 Importance of mitigation options

Optimal fertilisation rate (M1) and reduced organic fertilisation during winter season (M4) focus on reduced N input to cropland and can thereby reduce externalities of over-fertilisation in agricultural lands. M1 and M4 show a high negative CE of -251.2 and -475.8 €/tCO_2e, respectively. Scientific literature showed varying results for reduced fertiliser application i.e. Pellerin et al. (2013), Wang et al. (2014) and van den Pol-Dasselaar et al. (2013) also showed an extremely negative CE whereas Moran et al. (2011) and Schulte and Donnellan (2012) estimated a high CE. One reason is that assumptions about fertiliser reduction and the impact on crop yield are different within each study. In our study optimal fertilisation rate equals to crop N demand and no crop yield losses are expected. The low mitigation potential of 0.1 Mt CO_2e (M1) and 0.1 Mt CO_2e (M4) is surprising but justified by the Nitrate Directive that led to significant N-input reductions in agricultural soils. However, considering that the social cost of N excess outweighs the agricultural benefits by multiple times (Van Grinsven et al., 2013), a socially optimal fertilisation rate would be much lower as assumed in this study. This would increase the abatement potential as well as farmers' implementation costs tremendously.

Reduced tillage (M2) and cover crops (M3) are focussing mainly on increased C sequestration in agricultural soils. Although agricultural soils were significant C emitters until the end of the last century, this trend changed as indicated in our results i.e. SOC increases during the BAU scenario (Ciais et al., 2010; Lal, 2011). M2 and M3 approved that there is still a mitigation potential that comes with soil C sequestration. However, the mitigation potential is rather low as only a limited share of cropland for feed production was considered in this research. However, since M2 is available at negative cost of -63.4 €/tCO2e, policies should enforce this mitigation option. The estimated CE of M2 and M3 is in line with scientific literature (Moran et al., 2011; Pellerin et al., 2013; Schulte and Donnellan, 2012).

Dietary- lipids (M5), nitrate (M6), tannins (M7), probiotics (M8) and animal selection (M9) focus on enteric CH_4 reduction. Enteric CH_4 production was the largest GHG source in dairy production with a contribution of above 50% to total dairy sector's GHG emissions in 2020.

The CE of abatement by feed additives is not well represented in scientific literature and only recent studies reported CE's of some additives (Doreau et al., 2014; Pellerin et al., 2013; van Middelaar et al., 2014; Wang et al., 2014). Our results showed that the mitigation potentials are moderate i.e. between 0.44 – 1.35% reductions of dairy sector's total GHG emissions. Since total EU-15 dairy cows were divided to apply only a single additive per animal, implementation of only one feed additive to all dairy cows in the EU-15 dairy sector would increase the adoption potential and consequently the abatement potential strongly. In line with scientific literature, a high CE for M5 and M7 was estimated. Currently, it cannot be advised to implement these feed additives on a large scale. Regarding M5, the price of lipids increased strongly during the last decade. It is expected that this trend will continue until 2020 since biodiesel is prepared from rapeseeds amongst others and hence in direct competition for being a nutrition source (Balat, 2011). Alternatively, by-products of biodiesel generation could be used as feed addition as they contain lipids in highly concentrated form (Hristov et al., 2013). Further research needs to prove the effectiveness on enteric CH_4 reduction. Addition of unsaturated lipids to cow's diet could be beneficial to human health as they can replace saturated fats in milk and milk fat contributes largely to daily fat intake of humans in Europe (van Middelaar et al., 2014). This could also increase the milk price and hence decrease the CE. However, there is evidence that unsaturated fats could be saturated by microbes in the rumen and thus milk composition changes are small. Further, different fats can affect milk fat content differently (Patra, 2013). The CE of this mitigation option is thus dependent on the lipid used and could decrease if the fat content in the milk decreases and the farmer is paid based on milk fat content. Despite M6 having gained in popularity for effectively reducing enteric CH_4 emissions, this mitigation option is costly, does not act as a growth promoter and includes the risk of intoxication of the animal.

The high CE of M7 is driven by the costs of extraction of tannins and without large demand, production costs remain high. M8 shows a CE of -15 €/tCO_2e. Alternatively, tannin containing plants e.g. legumes could be introduced to pastures as these shows also impressive GHG reduction potentials (Archimède et al., 2011). Further, research should focus on the costs of this alternative. The CE of dietary probiotics is not well represented in scientific literature and the only study to present is Wang et al., 2014, which also stated a negative CE. However, M8 is a promising mitigation option, and it is advised to implement M8 on a large scale. Vellinga et al. (2011) showed that Dutch dairy farmers are being increasingly interested in replacing concentrates by feed additives; as they are driven by yield maximisation and awareness of the environmental impact of concentrate production. However, farmer's preferences for applying probiotics are limited (Klaus et al., 2014); mainly since there is a weak knowledge dissemination about its profitability. Animal selection (M9) focussing on CH_4 reduction and increased milk productivity is highly profitable with a CE of -339.8€/tCO_2e which is in line with scientific literature. This mitigation option shows a moderate mitigation potential of 0.7 Mt CO_2e while the adoption rate is rather low. To achieve a larger abatement potential, conventional breeding practices that focus only on increased cow yield, should be replaced by these focusing on different traits including enteric CH_4 reduction. Genomic selection allows to select animals accurately early in life based on genomic predictions for traits that are difficult or expensive to measure (Hayes et al., 2013). In this study it is assumed that cows are selected based on proxy traits for CH_4 production. However, there are currently no reports of genomic prediction for methane emission levels and thus reference data banks of several thousand animals needs to be generated to allow for accurate predictions (Hayes et al., 2013). There is a positive genetic correlation between residual feed intake and methane emissions as cows with lower residual feed intake have lower methane emissions (Haas et al., 2011). Therefore, reduced methane emissions could be achieved by selecting more efficient animals. Selection for residual feed intake results in cows with decreased DMI, increased feed conversation ratio and reduced CH_4 production while maintaining production levels (Basarab et al., 2013). Thereby, the farm area to produce feed could be decreased and that results in further GHG savings (Basarab et al., 2013). This mitigation option becomes even more interesting in light of increasing prices for feed e.g. grains and concentrates and could deliver even a higher CE in abating GHG emissions (Egger-Danner et al., 2015). However, large quantities of feed intake data on cows are required.

The comparison of MACC studies with our results revealed large differences in the estimated CE for the mitigation options. This is of concern for the MACC user. In order to avoid decision based on wrong predictions, MACC studies must be presented in a transparent way. This allows verification by a third party prior to policy decisions. Thereby, it can be ensured that complete information is used and if necessary that the MACC is updated in case of availability of better data. Further, policy maker should consider large heterogeneity of the agricultural sector and that estimates vary considering different farm types or regions. Therefore, prior to large scale implementation, selected farms could implement proposed mitigation options to understand the impacts in reality and be able to compare model results with reality.

6.4.2 Uncertainty of the results

It was attempted to assess uncertainty underlying this MACC exercise in two systematic steps; first, understanding the uncertainty involved in the input parameters and second assessing the uncertainty that comes with the measure's CE.

Similar to Eory et al. (2014), the uncertainty contribution of three input categories was assessed, i.e. baseline development of the dairy sector's activities, measure's implementation cost and measure's adoption potential. This assessment does not represent all input parameters for generating a MACC. Further, if considering the complete climate change feedback, even more input parameters trigger the total uncertainty i.e. atmospheric concentration change, weather changes, system impact, climate change impact on the individual (Smith and Stern, 2011). In this study, only a small part of this climate feedback chain was assessed and hence the contribution to total uncertainty may be under-represented, especially when considering that these input categories are interacting with each other. The total uncertainty of cost-negative abatement is 24.6%. The input category adoption potential shows the highest contribution to total uncertainty. This is a startling result since this category only consists of behavioural data on measure's uptake which is mostly dependent on expert judgement. Recently some studies have been published in the area of measure uptake potential (Klaus et al., 2014; Vellinga et al., 2011). Sánchez et al. (2014) showed that for farmers in Aragon (Spain), environmental awareness, access to technical advice and financial incentives are the main factors that limit adoption of mitigation options. Education related to technical training, environmental awareness and financial benefits of implementing mitigation options plays a critical role in increasing farmers' willingness to adopt mitigation options. Generally, younger farmers are more likely to adopt mitigation options compared to older farmers as they have higher awareness of environmental factors (Sánchez et al., 2014).

Therefore, education measures should target older farming population as a better informed farmer is more likely to adopt mitigation options. Education can be provided by local cooperatives, agricultural associations and research institutions. Financial incentives e.g. subsidies or monetary compensation are a main driver to increase adoption of mitigation options but these are limited by social and policy barriers (Prager and Posthumus, 2010). Farmers are likely to copy behaviour of other farmers driven due to an urge for conformity with other farmers. It is more likely that the behaviour of successful farms is copied to gain positive attention (Moran et al., 2013). This mechanism can be utilised to increase adoption rates of mitigation options if visual symbols that credit ’good farmer’ performance are used to advertise mitigation efforts by prestigious farms to than have a copying effect on neighbouring farmers (Moran et al., 2013). However, it remains important to further assess farmer’s preferences in a scientifically sound way for generating robust MACCs.

The second highest contribution to total uncertainty shows implementation costs of the mitigation options that consist only of economic data. This clearly shows that only the output of robust economic model should feed into the MACC construction to minimise contribution of this uncertainty source. In economic terms this means that a good data set should underlie the model, depending on the data availability and thus results in high quality predictions. However, in a scientific understanding a robust model can be validated with reality but this is not possible for economic models projecting the future.

The MC simulation revealed that the ranking of mitigation options is robust i.e. by considering the highest probability that a measure is available at a certain CE, the ranking remains equal to the initial MACC. This implies that a prioritisation of mitigation options based on the criteria of cost-effectiveness in abating GHG emissions is valid. However, measure’s CE is highly uncertain, except for M2, M3 and M6. These results further strengthen the recommendation for M2 and M3 being most prominent of the investigated mitigation options for large scale implementation in the European dairy sector. M4 and M9 are stable on the negative CE side, and hence our calculation revealed no risk of additional costs by implementing these measures.

Assessment of uncertainty contribution and uncertainty of the CE is an important step in evolving the MACC methodology and improving the robustness of the results. Since the MACC construction is dependent on research results of other studies, these studies should assess and report the pdfs of their results. Being able to utilise specific pdfs for a MC simulation generates crucial advantages as more justified definition of pdfs and hence a more robust uncertainty assessment can be applied. This is because same pdfs to all input parameters were attributed since quantitative information on pdfs for all input parameters was not available.

6.5 Conclusion

Despite of a high production efficiency in the EU-15 dairy sector in terms of GHG emissions per L milk and which is likely to even increase by 2020, there is an immediate need to cut GHG emissions and intensify the dairy sector in a more sustainable way. In this context, this chapter assessed the CE of 9 mitigation options applicable in the EU-15 dairy sector. It was shown that animal selection, reduced tillage and dietary probiotics are promising mitigation options that should be prioritised by policy makers. However, a comparison of our results with other MACC studies revealed that there are large differences in the estimated CE of the mitigation options. While this can be attributed to different focus and methodologies of these MACCs, our study also showed that there is a large uncertainty for most CE estimates in our study and probably also for other MACC studies. By applying equal pdfs to the model input variables, our MC simulation was found to show some limitations but the results revealed that MACC studies need to deal with uncertainties as not doing so may misguide mitigation policy decisions. This study also revealed the uncertainty contribution of several input variables that were segregated into three categories. The category of adoption potentials showed the highest contribution to uncertainty in our results thereby leading to the conclusion that more research is needed to improve the robustness of assumptions with respect to the adoption potential of different mitigation options.

Chapter 7 - Variability of marginal abatement cost estimates

7.1 Introduction

Over the last two decades, scientific literature has focused on estimating MAC in the agricultural sector for identifying cost-efficient mitigation options. Based on these results, the sector has been identified as potentially delivering large GHG abatement potentials that are available at low or negative costs.

Reviewing 77 studies that directly or indirectly estimated the CE of GHG abatement for mitigation options in the EU-25 agricultural sector, Bosello et al. (2005) revealed significant heterogeneity of modelling approaches for estimating MAC. More recently, Vermont and De Cara (2010) used a meta-regression analysis of 21 MACC studies to demonstrate that variability of abatement rates at different carbon prices is large and that study design e.g. study quality or geographical focus, are significant in explaining this variability. Considering the large variability in study design for ENG MACCs, it is likely that MAC estimates for any single mitigation option in the agricultural sector are also highly variable; hence there will be uncertainty related to the expected abatement potential and corresponding CE. Tol (2005) developed an interesting approach to quantify uncertainty of marginal damage cost (MDC). Based on a meta-analysis of 28 studies, the study reported pdfs of MDC estimates and visualised the influences of different study characteristics on the pdfs. The study reported a large variability of MDC estimates, thereby elaborating the uncertainties in results across the scientific literature. The variability regarding average MDC was largely determined by study quality and discount rate. Such an attempt has not been undertaken for ENG MACCs within the agricultural sector. Nevertheless, it could clarify uncertainty regarding MAC estimates, and thereby improving their role in informing policy decisions.

This chapter aims to quantify the variability of MAC estimates for eight mitigation options in the agricultural sector and to identify measures of highest cost-effectiveness in abating GHG emissions across the literature. The next section describes the meta-analysis on available ENG MACC literature and construction of the database. Section 2 shows the range of measures' CE estimates and the probability of these to be available at reference carbon prices. Section 3 discusses possible reasons for MAC variability and the urgency of communicating uncertainty to policy makers.

7.2 Data base generation

The selection of studies followed a systematic search using the keywords "agriculture", "marginal abatement", "costs" and "mitigation" on the main search engines, i.e. Thomson Reuters Web of Science, Google Scholar, AgEcon search and Science Direct. The period of search was not defined. Grey literature was included e.g. institutional reports and proceeding papers. Although grey literature may bias measure CE variability, a substantial number of ENG MACC studies are not published in peer-reviewed journals (Table 7.1). Initially 45 studies were selected (Table A.3) and these were shortlisted on the basis of the following criteria. First, the study must focus on the agricultural sector or a subsector. Second, main GHG emissions from agricultural production i.e. CH_4 and N_2O are considered. Third, studies report an economic abatement potential. Finally, studies present CE of GHG abatement for individual mitigation options. Since only MACCs based on ENG or hybrid approaches report these, SSM, CGE and PEM based MACCs were excluded. This criterion is essential for this review as the variability of MAC estimates for individual mitigation options across studies were assessed. Following these criteria, 19 studies were selected. Of these, 11 are institutional reports, 6 are peer-reviewed publications, one is a proceedings paper and another a doctoral dissertation (Table 7.1). Some of these studies only reported approximate CEs and abatement potentials and were illustrated diagrammatically. In these cases, the authors were contacted in order to acquire definite values. In cases of 'no response', the values from the diagrams were approximated (i.e. for Hasegawa and Matsuoka, 2010; Nauclér and Enkvist, 2009).

Table 7.1: Overview of studies included in the meta-analysis. Studies were categorised in different clusters if published by same authors and based on a shared database.

Author(s)	Year	Document	MACC type*	Focus	Cluster
Amann et al.	2008	Institutional report	Hybrid	EU-27 countries	1
Bates	2001	Institutional report	ENG	EU-15 countries	2
Bates	2009	Institutional report	ENG	EU-27 countries	2
Branca et al.	2012	Proceedings paper	ENG	Malawi, 2 regions	3
de Oliveira Silva et al.	2015	Peer-reviewed paper	ENG	Brasil, one region	4
Gouvello	2010	Institutional report	ENG	Brasil	5
Graus et al.	2004	Institutional report	ENG	Global	6
Höglund-Isaksson	2012	Peer-reviewed paper	Hybrid	Global	7
Kahil and Albiac	2013	Peer-reviewed paper	ENG	Spain, one region	8
Koslowski et al.	2015	PhD thesis	ENG	EU-15	9
Moran et al.	2011	Peer-reviewed paper	ENG	United Kingdom	10
Moran et al.	2008	Institutional report	ENG	United Kingdom	10
Nauclér and Enkvist	2009	Institutional report	ENG	Global	11
Pellerin et al.	2013	Institutional report	ENG	France	12
Schulte and Donnellan	2012	Institutional report	ENG	Ireland	13
Smith et al.	2008	Peer-reviewed paper	ENG	Global	14
USEPA	2006	Institutional report	ENG	Global	15
van den Pol-Dasselaar et al.	2013	Institutional report	ENG	Nether-lands	16
Wang et al.	2014	Peer-reviewed paper	ENG	China	9

*In this review two different MACC approaches were included, i.e. ENG and hybrid MACC approaches.

Some studies reported several MACCs based on various time horizons (e.g. Moran, 2008) or based on different GHG accounting methodologies e.g. LCA (Schulte and Donnellan, 2012; Pellerin et al., 2013). First, several MACCs from these studies were extracted and at a later step, LCA based MACCs were excluded from the review. Since some studies presented individual MACCs for different regions or countries (see Graus et al., 2004; US EPA, 2006; Smith et al., 2008), one single MACC for each country or region were extracted. Based on the target country, the MACCs were assigned to a particular region variable i.e. Europe, North-America, Asia, Global and Other (including South-America and Africa). The region code 'Global' was assigned to MACCs that are not country-specific and could not be separated to one of the region codes (see Graus et al., 2004; Smith et al., 2008). If a MACC represents global agriculture but reported specific to countries or regions, this MACC was broken down to several MACCs. Based on this data set, the CE for individual mitigation options was extracted. To each MAC estimate, several study characteristics were assigned (Table 7.1 and Table 7.2). These included: i) the quality of the study i.e. peer-reviewed or not, ii) geographical focus, iii) economic sector, iv) simulated baseline horizon, v) GHG emission considered (i.e. CH_4, N_2O, CO_2 and SOC), vi) whether the study accounted for interaction between the mitigation options and vii) GWP used for conversion into CO_2e. These study characteristics will later help in identifying potential sources of variability in MAC estimates. Further, studies were categorised that are published by same authors or compiled common databases into one cluster; resulting in 16 clusters (note that some clusters can also include only one study, Table 7.1).

Some studies reported for some measures in different regions, identical CE estimates (see Graus et al., 2004). Since this might indicate a potential shortcoming of the modelling methodology and thus could bias the database, all duplicates where more than 2 occurring in the study were deleted and measures' region was set to 'Global'. To generate a consistent and comparable database, the data points were normalised. For the CE and unit costs, the prices were converted to $€_{2011}$ for the EU using the Consumer Price Indices and Purchasing Power Parities from OECD (http://stats.oecd.org). Additionally, all measures were deleted that showed a CE above 2500$€_{2011}$/t CO_2e or below -2500$€_{2011}$/t CO_2e. Only few data points are reported with extreme CEs. For instance, Moran et al. (2008) claimed for the mitigation option 'reduced fertiliser' a CE of 23708 $€_{2011}$/tCO_2e. Such an outlier would affect this analysis strongly and there is a high probability that these estimates are subject to shortcoming in prediction. After generating this final database, the mitigation options were selected by taking into account the following criteria: i) the mitigation option must include

25 data points and ii) these data points must arise from at least three different studies (to have sufficient data to be able to fit statistical models). Selected mitigation options are: i) reduced amount of mineral fertiliser application (REDFERT; 109 observations), ii) multiple application of fertiliser (SPLITFERT; 26 observations), iii) improved timing of fertilisation i.e. matching the time of the fertiliser application with the demand of the plants (TIMEFERT; 32 observations), iv) transition to no-tillage cropping systems (NOTILL, 26 observations), v) on field application of soil nitrification inhibitors (NITR, 57 observations), vi) on-farm anaerobic digestion with electricity generation (AD-E; 79 observations), vii) on-farm anaerobic digestion without electricity generation (AD-H; 51 observations) and viii) centralised anaerobic digestion in large scale plants (CAD; 79 observations). A detailed overview of the raw data can be obtained from the Appendix 3 (Table A.4).

It is important to mention that assumptions behind single mitigation options vary considerably and may hence lead to variations of measures' CE estimates. For instance, in the case of REDFERT, some studies state a fixed fertiliser reduction e.g. by 50Kg /ha, while others assume an optimal fertilisation depending on the biophysical conditions. Since studies only report fixed values for measure's CE, it was not possible to harmonise the data in this regard. The final dataset includes 460 data points collected from 19 studies (Table 7.1 and Table 7.2).

Table 7.2: Parameter characteristics of studies included in the meta-analysis.

Study	Sector	Projection horizon	GWP	Interaction	GHG included			
					CH_4	N_2O	CO_2	SOC
Amann et al. (2008)	Whole economy	2005 - 2020	IPCC (1996)	yes	+	+	-	-
Bates (2001)	Agriculture	1990 - 2010	IPCC (1996)	no	+	+	-	-
Bates et al. (2009)	Agriculture	2005 - 2020	IPCC (1996)	no	+	+	-	-
Branca et al. (2012)	Agriculture	2010 - 2015	IPCC (2007)	no	+	+	-	+
de Oliveira Silva et al. (2015)	Livestock sector	2006 - 2030	IPCC (2007)	yes	+	+	+	+
Gouvello (2010)	Whole economy	2010 - 2020	IPCC (1996)	no	+	+	+	+
Graus et al. (2004)	Agriculture	2004 - 2020	IPCC (1996)	no	+	+	-	-
Höglund-Isaksson (2012)	Whole economy	2005 - 2030	IPCC (1996)	yes	+	+	-	-
Kahil and Albiac (2013)	Agriculture	2008 -2012	IPCC (1996)	no	+	+	-	-
Koslowski (2015)	Dairy sector	2009 - 2020	IPCC (2007)	yes	+	+	-	+
Nauclér and Enkvist (2009)	Whole economy	2008 - 2030	IPCC (1996)	no	+	+	+	+
Moran et al. (2008)	Agriculture	2006 - 2012/2017	IPCC (1996)	yes	+	+	+	+
Moran et al. (2011)	Agriculture	2006 - 2022	IPCC (1996)	yes	+	+	+	+
Pellerin et al. (2013)	Agriculture	2010 - 2030	IPCC (2007)	yes	+	+	+	+
Schulte and Donnellan (2012)	Agriculture	2010 - 2020	IPCC (2007)	no	+	+	+	-
Smith et al. (2008)	Agriculture	2008 -2030	IPCC (2007)	no	+	+	+	+
USEPA (2006)	Whole economy	2000 - 2020	IPCC (1996)	no	+	+	-	+
van den Pol-Dasselaar et al. (2013)	Dairy sector	2013	IPCC (2007)	no	+	+	+	+
Wang et al. (2014)	Agriculture	2010 - 2020	IPCC (1996)	yes	+	+	+	+

7.3 Methodologies

7.3.1 Description of kernel density estimation

The simplest way of visualising the frequency distribution of a dataset is a histogram that consists of several uniform blocks explaining the entire range of the data set. The height of these blocks is determined by a number of values that fall into one block. The shape of a histogram may therefore strongly vary if different intervals (width of the blocks) are chosen, and this may lead to different shapes of the density distribution. A more realistic idea of the distribution of values can be obtained by kernel density estimation (KDE; Silverman, 1986). This is advantageous in many ways as compared to presenting the mean and standard deviation of the raw data set. This is because it gives information on different phenomena of the data set e.g. asymmetry, non-normality and multi-modality (Tortosa-Ausina, 2002).

KDE is a non-parametric tool that was widely applied in the 1990s and has since undergone a large development (Härdle et al., 2004); hence it can be assumed that most issues in terms of univariate data have been solved. For multivariate data, this tool might have some disadvantages compared to other approaches in filling data gaps (Lokupitiya et al., 2006). The rationale behind Kernel smoothing is that a certain value within a limited data set contains information about its neighbouring values. The smoothing aims to utilise the information given in a data set and reduce random variability being caused by a finite sample size.

For univariate data {y1, y2, …, yN}, the formula for the kernel density estimator is as follows:

$$\hat{f}(x) = \frac{1}{Nh}\sum_{n=1}^{\mathrm{N}} \mathrm{K}\left(\frac{x - x_n}{h}\right) \tag{4}$$

Where N is the number of data points for a single mitigation option, K is the kernel selection function, h is the bandwidth or smoothing parameter and x is a random variable.

The kernel selection function attributes to each value in a given data set a kernel i.e. a function available in different forms e.g. Gaussian, triangular, uniform. The smoothing parameter determines the width of the kernels and the height of the kernels is equal in case of unweighted data, and for weighted data the kernel height changes according to the weight specified. Figure 7.1 shows the principle of KDE.

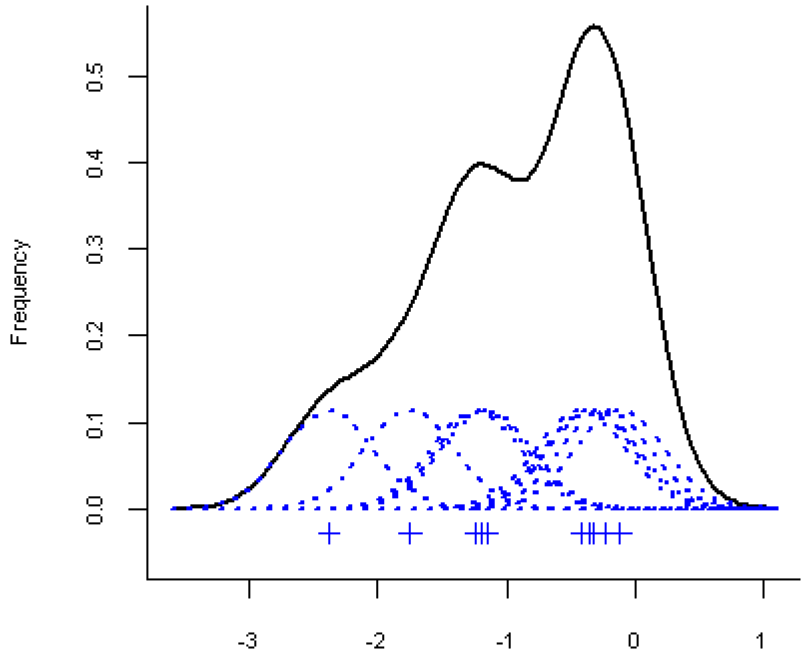

Figure 7.1: Example of the basic principle of kernel density estimation. The blue crosses indicate single values of a given variable and the dotted lines above are the Gaussian kernel function that are attributed to each of these points. The sum of the kernel functions provides an estimate of the probability distribution function of the variable.
Source: altered from Härdle (2004).

The bandwidth selection is the most important factor for KDE as it controls the smoothness or roughness of the density estimate. In case of choosing a too small bandwidth, there arises the phenomena of under-smoothing the density function, thereby leading to the generation of a vast number of bumps which implies an extremely variable distribution; whereas a too high bandwidth causes the phenomena of over-smoothing and thereby strongly reduces the variation (Tortosa-Ausina, 2002; Figure 7.2). As recommended by Jones et al.(1996), a so called “second generation” kernel density estimator should be used. These are superior to the “first generation” methods in terms of smoothing the data. “Second generation” methods use an automatic data-based bandwidth selection that results in the best trade-off between under- and over smoothing, or in other words equality between “bias” and “variance” (Jones et al., 1996). The bandwidth being chosen by the plugin-selector increases in accuracy as the size of the dataset increases (Matthiopoulos, 2003).

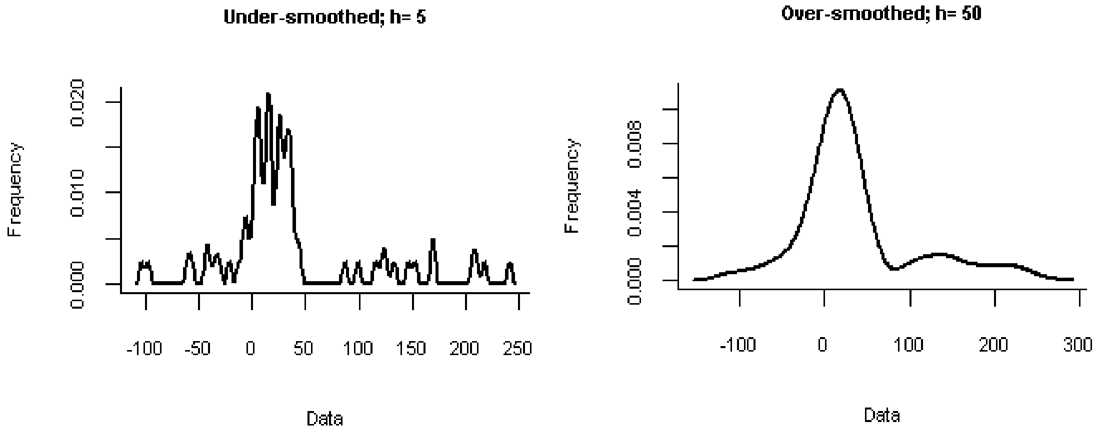

Figure 7.2: Example of under-smoothing (left side) and over-smoothing (right side). The bandwidth (h) is stated above the graphs.

However, there are situations in which not every data point is equal in weight. For instance, in case of multiple data points reported by one study, these data points have lower weighting than a study reporting a single value since it is likely that multiple observations correlate with each other and may therefore bias the data set (Nelson and Kennedy, 2009). Therefore, in this chapter for each observation a weight was attached to account for a more realistic contribution to the dataset (Gisbert, 2003). Accounting for unequal contribution of different observations to the whole data set leads to the following formula:

$$\hat{f}(x) = \frac{1}{\sum n\,\omega_n Nh} \sum_{n=1}^{N} \omega_n K\left(\frac{x - x_n}{h}\right) \qquad (5)$$

Where ω_n is the weighting factor for each observation. According to the value of the weights, the height of the kernels for each observation will increase or decrease.

So far the use of a fixed bandwidth estimator was discussed where each kernel has the same bandwidth. Adaptive KDE allows selecting local bandwidths i.e. the bandwidth can vary for each data point (Silverman, 1986). However, the impact of adaptive kernel density estimator on the pdfs is small as compared to weighted KDE and may not show significant improvements in predicting the pdf (Gisbert, 2003).

7.3.2 Kernel density estimation for generating probability distributions

In this chapter, KDE was used to visualise the range of reported CEs for single mitigation options. A fixed bandwidth kernel density estimator was utilised based on the package “Kernel smoothing” (ks) in R (Duong, 2007). A copy of the code that has been used for the KDE is included in Appendix 4. A normal distribution (i.e. Gaussian form) was used for the kernels as it is standard in most software packages. The reason is simply that Gaussian kernels allow generating a smooth pdf rather than for instance having sharp cut-offs by using uniform kernels. The bandwidth selection is based on the default plugin-selector “hpi” in the ks package (based on Wand and Jones, 1994). This plugin-selector for one dimensional data uses a direct plug-in methodology that includes kernel estimates for unknown quantities that appear in the formula for the asymptotically optimal bandwidths. The asymptotically optimal bandwidth is the expression in which the selection of the heights of the kernels minimises the mean squared error (used for measuring precision; Wand and Jones, 1995).

As part of a sensitivity analysis, the observations were weighted for one mitigation option from the same cluster by their relative contribution to total observation number of that cluster i.e. total weight for each cluster is equal. KDE was then run for the unweighted and weighted dataset. In a second step of the sensitivity analysis, the variables study quality and location (MAC estimates that are reported for Europe) were considered. The KDE was then re-run only for the weighted dataset while accounting for these variables.

For ‘study quality’, it assumed that study quality impacts the variance of MAC estimates. Here, the kernel density was estimated with data points that are only reported in grey literature. KDE based on ‘peer-reviewed’ data points were not suitable since too few data points are reported for some mitigation options. The fact that non-peer-reviewed studies report many data points is certainly a drawback in terms of robustness of the MACC results. For ‘location’, it was assumed that the heterogeneity within the agricultural sector in Europe is smaller as compared to the global agricultural sector and this may impact variance of measure’s CE.

The probability of a mitigation option being available at a certain carbon price was derived based on the individual cdf. Reference carbon prices were used that are usually stated in literature in the climate change mitigation debate i.e. 0, 10, 25, 50 and 100€$_{2011}$/tCO_2e.

Finally, the mitigation options were ranked based on the probability of the mitigation option being below 0€$_{2011}$/tCO_2e (for AD-E in the cluster based ranking, there was an exemption since this measure shows a higher probability of being available at 10 and 20€$_{2011}$/tCO_2e as compared to TIMEFERT). As part of our sensitivity analysis, two rankings were obtained, being derived from cdfs for the unweighted and the weighted dataset.

7.4 Results

7.4.1 Kernel density estimation and variability of the data set

It is evident that variability of MAC estimates for the eight mitigation options is enormous, with the largest range for REDFERT and CAD (Table 7.3 and Table A.4). Figure 7.3Aa – Ha give a clear indication of this range. Although some pdfs appear to be normally distributed, there are several extreme CE estimates reported and thereby leading to slightly under-smoothed pdfs.

Table 7.3: Probability characteristics of the cost-effectiveness of abatement (€$_{2011}$/tCO_2e) for each mitigation option, based on kernel density estimation of 4 different datasets.

Mitigation option	Data set	Mode	Mean	Median	5%	10%	90%	95%
REDFERT	Unweighted	6	44	19	-620	-255	274	629
	Weighted	-42	154	-10	-266	-216	686	1768
	Study quality	-22	262	33	-82	-60	1721	2463
	Location	-39	236	9	-271	-244	1730	2470
SPLITFERT	Unweighted	3	57	9	-95	-76	298	479
	Weighted	0	62	36	-71	-49	165	212
	Study quality	0	62	36	-71	-49	165	212
	Location	1	61	37	-24	-16	142	150
TIMEFERT	Unweighted	42	-4	38	-167	-135	69	82
	Weighted	-92	-163	-98	-497	-491	49	61
	Study quality	-92	-47	-89	-166	-130	54	67
	Location	-92	-163	-98	-497	-491	49	61
NOTILL	Unweighted	-71	-64	-63	-196	-157	26	60
	Weighted	-2	-33	-19	-159	-117	30	45
	Study quality	-2	-33	-19	-159	-117	30	45
	Location	28	-43	-1	-356	-315	134	168
NITR	Unweighted	81	97	79	-13	2	204	307
	Weighted	48	103	66	-15	-1	243	388
	Study quality	50	81	61	-15	-2	201	232
	Location	59	108	67	-4	5	249	396
AD-E	Unweighted	19	43	19	-84	-40	160	216
	Weighted	1	134	25	-29	-14	711	780
	Study quality	12	133	24	-46	-13	774	784
	Location	10	159	31	-32	-16	774	788
AD-H	Unweighted	32	168	71	-6	6	445	875
	Weighted	32	182	95	-1	12	427	752
	Study quality	32	182	95	-1	12	427	752
	Location	101	194	125	20	42	415	437
CAD	Unweighted	7	569	308	-117	-67	2034	2204
	Weighted	5	238	49	-151	-111	682	1931
	Study quality	6	259	58	-161	-117	889	1983
	Location	17	294	102	-253	-183	1343	2029

By simply estimating the average for the unweighted data, TIMEFERT and NOTILL are reported to be cost-negative and hence are the most cost-efficient mitigation options (Table 7.3). The disadvantage of using unweighted data is that those studies reporting multiple data points are given undue weight. The mean of the weighted dataset and the data sets that accounts for the variables 'study quality' and 'location' drastically changed for half of the mitigation options (i.e. REDFERT, TIMFERT, AD-E and CAD) at first view, but this effect is far less if the large range of the data points is considered.

Considering the mode of the observations, REDFERT is also assumed to be cost-negative, although not for the unweighted data (Table 7.3). The mode of measure CE differs strongly from the mean and is in most cases lower, except for TIMEFERT and NOTILL (Table 7.3). This indicates not only a large uncertainty in the estimates, but also that the mitigation options are likely to be more cost-efficient than indicated by the mean. However, considering only one single value for measure CE may under-represent the large range of reported estimates.

The cdfs show the estimated probabilities that a measure is available at a reference carbon price (Figure 7.3Ab – Hb). Including weights have a large impact on these probabilities, particularly for REDFERT, SPLITFERT, TIMEFERT, AD-E and CAD (Figure 7.3Ab, Bb, Cb, Gb and Hb).

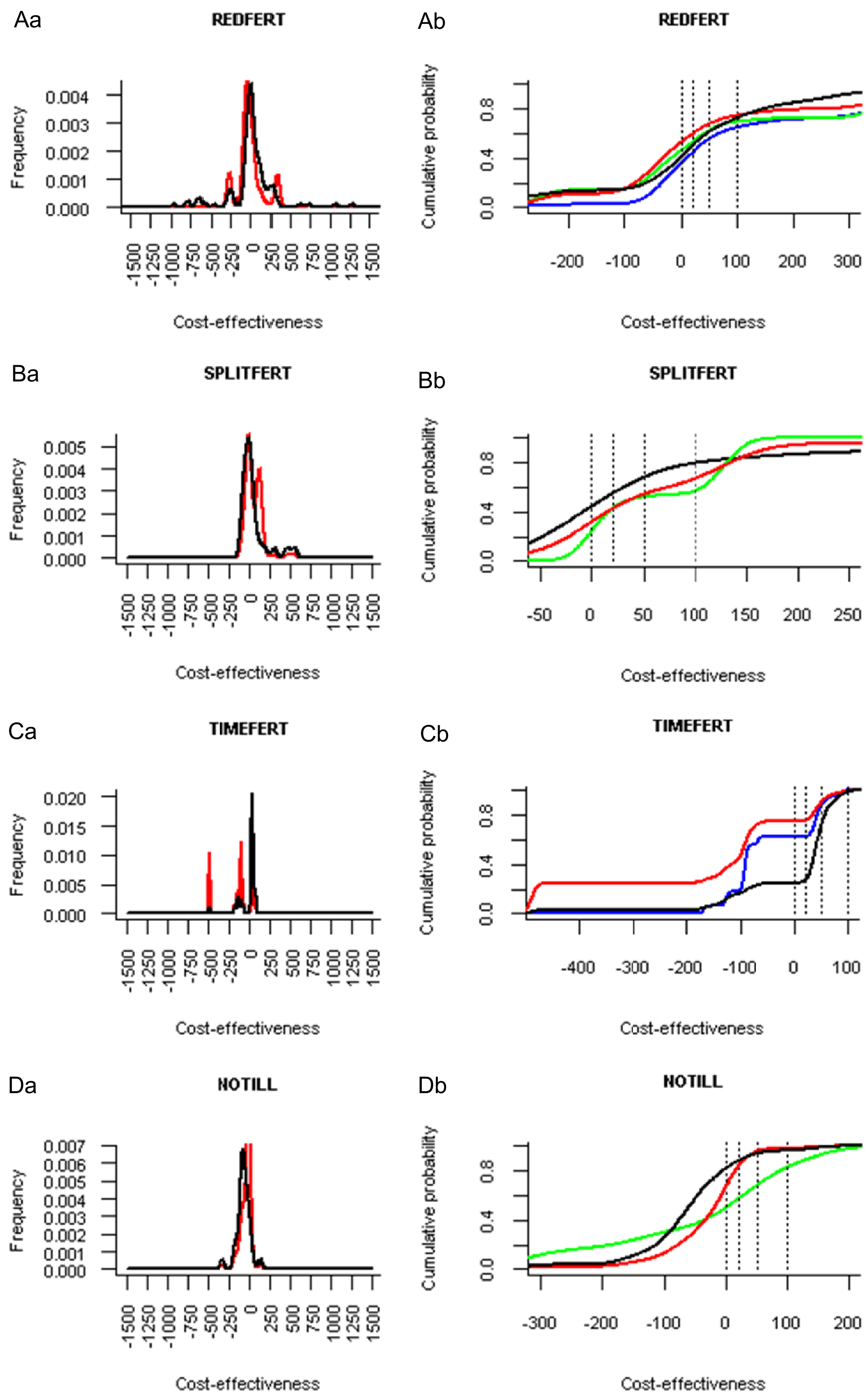

Figure 7.3 (Aa – Hb): Probability distribution functions and cumulative distribution function. On the left are the pdf derived by KDE for 8 mitigation options, on the right are the cdfs derived by KDE for each mitigation option. Left: the black and red line represents the unweighted and weighted dataset, respectively. For the mitigation options “REDFERT” and “CAD”, the full range of the pdf is not shown in the graphs.

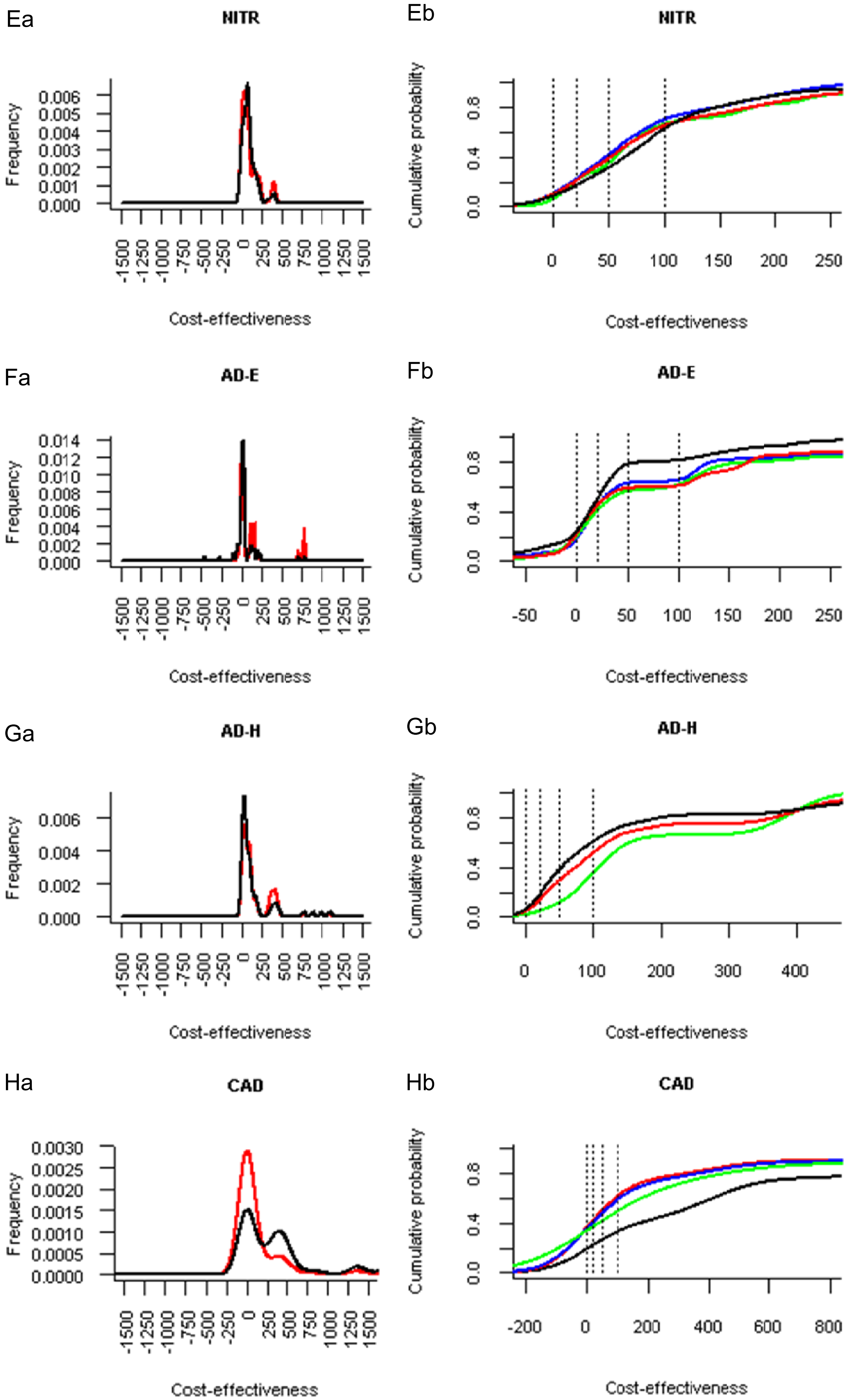

Figure 25: continued. Right: the colours of the different lines are explained as follows: i) black for the unweighted data set, ii) red for the weighted data set, iii) blue for the variable 'study quality' and iv) green for the variable 'location'. The dashed vertical lines indicate the carbon prices 0, 20, 50 and 100€/tCO_2e (from left to right). In some graphs different coloured lines overlay in case of identical data points. This is the case for the red and blue line in Figure 26Bb, Db and GB and for the red and green line in Figure Cb.

Since ‘study quality’ and ‘location’ are based on weighted data, these were not compared to the unweighted data. Selection only by ‘study quality’ shows a certain pattern of altering the cumulative probability i.e. REDFERT, TIMEFERT, NOTILL, AD-H have a lower likelihood of being available at reference carbon prices as compared to weighted data (Figure 7.3Ab, Cb, Db and Gb); indicating that peer-reviewed publications report lower CE for these mitigation options. However, due to a low number of “peer-reviewed” observations, this is not very robust evidence. ‘Location’ shows a lower probability for TIMEFERT, NOTILL, AD-H and CAD being available at the reference carbon prices (Figure 7.3Cb, Db, Gb and Hb). These mitigation options might therefore be more costly in Europe. The range of CE estimates differs strongly as compared to weighted data points, but data based on ‘study quality’ and ‘location’ have a similar large variability and in some cases even larger; hence these variables do not reduce variability of measure’s CE.

Figure 7.4 shows a ranking of mitigation options based on cdfs from unweighted and weighted data. The mitigation options can be grouped into three different categories that focus on: i) reduced soil N_2O emissions (TIMEFERT, REDFERT, SPLITFERT and NITR), ii) reduction of N_2O and CH_4 from manure storage (AD-E, CAD and AD-H) and iii) increasing C soil storage (NOTILL). For soil N_2O measures, TIMEFERT and REFDERT ranked highest i.e. 75% and 53% probability of CE < 0€/tCO_2e, respectively. NITR is the least favourable in this category and shows a probability of being 35% above 100€/tCO_2e. The manure management mitigation options are less cost-effective compared to the soil N_2O measures and are in all cases available at very high prices. CAD shows the highest CE in this category with a probability of 36% being below 0€/tCO_2e. AD-H clearly shows the lowest CE amongst all measures with 48% probability being available at high carbon prices. NOTILL displays a likelihood of being 66% below 0€/tCO_2e and is therefore alongside TIMEFERT and REDFERT, the most cost-efficient mitigation option.

As stated earlier, cdfs based weighted data points show strong differences to those based on unweighted data; hence it is not surprising that the ranking differs for some mitigation options if they are based on different datasets (Figure 7.4). Considering the unweighted dataset, TIMEFERT becomes less likely to be cost-efficient, and its ranking as ‘best’ soil N_2O mitigation option is replaced by SPLITFERT. Similar to this, AD-E replaces CAD as most cost-efficient manure mitigation option and CAD becomes in turn a high-cost measure with 67% probability being available above 100€/tCO_2e. These are striking results and must be considered when interpreting these rankings. However, there are also some similarities in both rankings i.e. NOTILL and REDFERT are the most cost-efficient and NITR and AD-H are the least favourable mitigation options.

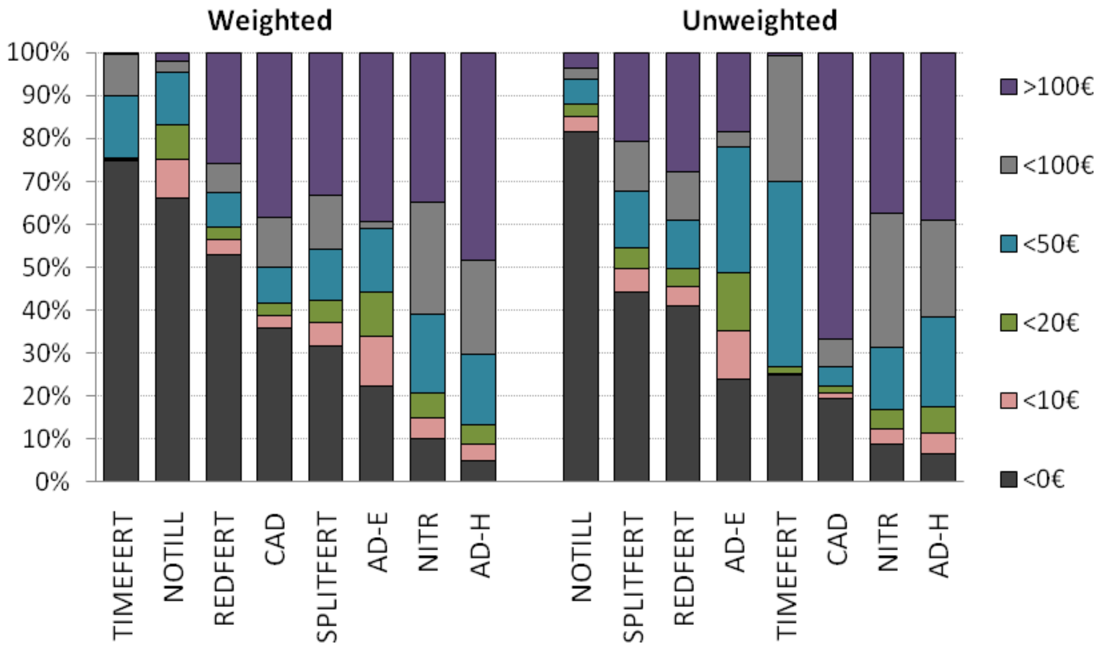

Figure 7.4: Ranking of mitigation options based on cumulative densities. The ranking is from left to right with measures with the highest probability of being cost-negative on the left side; for equity weighted KDE (left) and raw data KDE (right).

7.5 Discussion

7.5.1 Study design and their impact on marginal abatement cost

The results from our meta-analysis show that the 19 MACC studies have large variation in model design (Table 7.1 and Table 7.2), and this can significantly affect MAC estimates for single mitigation options (Vermont and De Cara, 2010). The following sections address the reasons why study design could affect MAC estimates but there are also other factors affecting MACs variability e.g. robustness of the modelling exercise, institutional and scientific bias in the scientific process, divergent information sources, reliability and availability. Since some of these potential biases can be associated to the variable study quality, it was tried to elaborate the effect of study quality on MACs in our results section.

7.5.1.1 Geographical focus

MACCs were developed for various regions (Table 7.1). Since agricultural systems vary considerably across the globe in terms of e.g. mechanisation levels, production intensities, biophysical settings and management implementation of mitigation options in different regions could lead to varying GHG abatement potentials and implementation costs. Smith et al. (2008) demonstrated variations in technical abatement potentials of various mitigation options across agricultural systems in different regions of the world. This shows that geographical focus can affect MAC estimates strongly, also in terms of implementation costs. This is consistent with our results that MACCs focusing on Europe showed strongly different mean and mode of MACs for several mitigations options. The user should therefore only utilise MACCs that are specific to the region of action as data from other regions can be misleading.

7.5.1.2 Geographic and economic resolution

MACC studies may focus either on regions within countries, whole countries, continents or even the globe (Table 7.1). Studies with a wider focus could have a lower geographical resolution and may thus simulate heterogeneous agricultural systems less accurately as compared to studies with a smaller geographical focus. Similarly, MACC studies can also target different economic sectors i.e. agricultural subsectors (dairy sector), total agricultural sector or all polluting economic sectors (Table 7.2). For instance, Smith (2012) claimed that abatement potentials of mitigation options differ significantly within one country, and this is also true for measures' implementation costs and adoption potentials; hence study resolution can have a crucial impact on MACs with higher detail levels, leading to more robust MAC estimations than studies with lower detail levels. Furthermore, focus on agricultural subsectors e.g. the dairy sector represents MACs that are only accurate for this subsector. For instance, if cropland measures are considered for this subsector, evaluating these mitigation options can lead to under-representation of their abatement potentials compared to application on total UAA. Additionally, the economic structure of dairy farms compared to farms of other agricultural subsectors differs. This may change implementation cost and abatement potentials of mitigation options and hence MACCs. The MACC user should be aware of this and use information of several MACCs focussing on the same region. Ideally, a preliminary MACC identifies cost-efficient mitigation options on large scale e.g. for the EU or economy wide. In a second step these results will be verified for region or sector specific MACCs particularly for those regions or sectors that are strongly different to average production systems in that region. For this the same MACC could be used but updated by region or sector specific data to understand on which regions the MACC user should focus to implement the mitigation options.

7.5.1.3 Accounting for different GHGs

Different MACC studies account for different GHGs (Table 7.2). While some consider only one GHG, others include all key GHGs from agricultural activities (i.e. CH_4, N_2O, CO_2), C sink through LUC and afforestation and offsets through renewable energy systems. It is likely that mitigation options affect more than one GHG source and hence focus on only part of their abatement potential can drastically affect MACs.

7.5.1.4 GHG accounting methodologies

Methodologies to estimate baseline GHG emissions and abatement potentials vary considerably amongst MACC studies. Studies use either methodologies based on GHG inventories, IPCC or their specific model solution. As shown by Durandeau et al. (2010), using the economic model AROPAj resulted in significantly lower N_2O emissions as compared to the IPCC method. In this regard, the methodology of GHG emission accounting can affect MACCs.

7.5.1.5 GHG accounting boundaries

Most studies account for the effect of mitigation options only for GHG emissions within defined geographical boundaries and in a ‘cradle to gate’ approach in which GHG emission outside the farm gate are not accounted to total GHG budget. The reason is simple. MACCs are tools to inform policy makers about CE in meeting national binding GHG reduction targets and based on GHG inventories for quantifying GHG reduction. These inventories exclude GHG emissions outside national boundaries and separate GHG emissions by economic sectors. GHG emissions outside the farm gate are related to other sectors e.g. transport or other countries despite being caused by agricultural production. Considering abatement potentials outside geographical and sectoral boundaries e.g. in a LCA will either increase or decrease the abatement potentials of mitigation options that change production inputs (e.g. feed and mineral fertiliser) and services that are produced or occur outside these GHG accounting boundaries (O'Brien et al., 2012, 2014). Consequently, variations in GHG accounting methodologies lead to different MAC estimates.

7.5.1.6 Application potential

The applicability of mitigation measures during the baseline scenario determines the abatement potential and hence strongly affects the MACs. Although studies use various approaches to determine measure's adoption potentials, the methodology is not clearly reported in the MACC literature. Two methodologies can be clearly differentiated for measure's adoption during the baseline. It can be either a linear adoption (e.g. Moran et al., 2011) or adoption in a sigmoid shaped form to simulate diffusion of new technologies with few early adopters in the beginning, rapid increase of adoption rate and few adopters at the end (e.g. Pellerin et al., 2013). However, since MACCs are usually reported for the last year of the project horizon, only the final year adoption is of importance for the final MAC estimate. As shown in chapter 6, the adoption potentials for each mitigation option contribute to overall uncertainty of the MACC exercise largely. Since the adoption rate is mostly based on expert judgement as model prediction is mostly not feasible this variable can be subject to high uncertainties. The estimates of potential adoption vary strongly across studies for single mitigation options and consequently lead to variations in MACCs. However, the application potential of single mitigation options is rarely reported in the MACC literature. More research is required to predict farmers' behaviour and their attitude towards implementation of mitigation options. As discussed previously, science is moving fast in this field but more efforts are required to fill this gap in scientific knowledge.

7.5.1.7 Projection horizon

The simulated time horizon varies considerably between studies i.e. it can be either the current or a 25 years' future point of time (Table 7.2). Unpredictability about technological change, LUC and agricultural, economic and climate disasters make MAC estimations over an extended period subject to uncertainty and contributes to the variability of MAC estimates. Further, discounting future cost and benefits is a problematic issue. The discount rate is an extremely uncertain parameter and uncertainty accumulates exponentially over time (Pindyck, 2007); making discounted cash flows, especially over long periods, highly uncertain. Uncertainties may arise from predicting the economic growth and hence the interest rates in future. For instance, unpredictability of future disasters that lead to changes in economic growth makes it impossible to find the correct discount rate; in this case, any non-zero discount rate would be wrong. A simple example shows the tremendous impact of the discount rate on the present value if a future benefit of 1000€ in 25 years is expected. The discount rates of 3% and 5% are commonly applied to long-term environmental investments but applying these discount rates would lead to present values of 478€ or 295€, respectively; a difference of 18% compared to the future benefit. For instance, the Stern review was criticised for either a too high or low discount rate which led to a misrepresentation of the economic impact of climate change (Pidgeon and Fischhoff, 2011). Despite this critical input to MAC estimations, the applied discount rate is rarely reported in MACC literature. Model-based MACCs are better equipped for long-term predictions as they integrate market feedbacks and to a certain degree evolvement of markets. However, the precision of the underlying production functions determines the certainty of the future predictions. For MACC users, the long-term effect of mitigation options would be desirable information. Contrary to e.g. climate projections by the IPCC, the economic component of MACCs is subject to many influencing factors that can happen at any given time. Therefore uncertainty increases substantially with a longer time horizon.

7.5.1.8 Global warming potentials

Usually, non-CO_2 GHGs are converted to CO_2e by using the GWP. Since CH_4 and N_2O contribute largely to total agricultural GHG emissions, the choice of the conversion factor has a significant impact on the abatement potentials expressed in CO_2e and hence MACs

(Reisinger et al., 2013). In our review, all 19 studies used GWPs published by either the IPCC report in 1996 or 2007; Table 7.2). For CH_4, the GWP over a 100 year time-frame changed considerably between the IPCC reports in 1996, 2001, 2007 and 2013 i.e. a GWP of 21, 23, 25 and 28, respectively (Houghton et al., 1996; Ramaswamy et al., 2001; Forster et al., 2007; Myhre et al., 2013;). Contrary to this, the GWP of N_2O remained relatively constant i.e. an decrease from 310 to 298 between 1996 and 2013. The GWP changed as results of increasing scientific understanding of the lifetime and energy absorption and changing atmospheric concentrations of GHGs that affects the efficiency of radiation absorption. Further, the GWP of CO_2 remains one and as GWPs of other GHGs are related, changes of the global warming impacts of CO_2 also affect GWP of other GHGs. Hence, GWP estimates will also change in future. The strong GWP increase of CH_4 is partly reasoned by consideration of indirect effects of methane in the atmosphere on other radiatively active substances as ozone in newer assessment reports (Howarth, 2014). GWPs are usually expressed in 100 year horizons. Considering the short atmospheric lifetime of 12 years, the GWP for CH_4 at a 20 year horizon would be largely higher and this has strong implications on the actual global warming impacts of CH_4 compared to CO_2 (Howarth, 2014). It is generally advisable to use the newest GWP reported by the IPCC as these reflect current understanding and development most accurately. And this is also common practice for the GHG inventories. However, it was not possible to convert the estimates in this study as GHG reduction were often reported in CO_2e.

7.5.1.9 Interaction between mitigation options

Some mitigation options interact if they operate at a farm simultaneously. For instance, GHG reduction from reduced fertilisation can be counteracted by a longer grazing period if the manure inputs to the pasture are increased. As discussed in Appendix 1, SSM, CGE, PEM and hybrid models account for interactions automatically, but ENG models need to assess interaction between measures manually. ENG MACC studies developed various ways to cope with interaction: i) applying interaction factors i.e. the abatement potential of mitigation option B is multiplied with an interaction factor that describes the interaction between mitigation option A and B if both are implemented simultaneously (see Moran et al., 2008). ii) implementing of mitigation measures that are adopted on an equal share of total livestock or cropland. Here, application of measure A does not coincide with application of measure B (Wang et al., 2014). iii) a hybrid of i) and ii) i.e. considering interaction for sub-measures within one main mitigation option category e.g. fertiliser application, energy savings, methanisation, followed by accounting of interaction between main measures by adjusting their potential applicability, with favourite mitigation options being implemented first (see Pellerin et al., 2013). While application of interaction factors is subject to uncertainty since simulation of potential interactions are an uncertain model exercise, not all ENG MACC studies account for interaction at all (Table 7.2). However, applying one of the three interaction accounting methodologies or none affect measure's CE strongly as application potentials and abatement potentials differ depending on the selected interaction scenario (for an example, see Moran et al., 2008).

7.5.2 Communication of MACC uncertainty

Current MAC studies suggest that mitigation options are available at a certain carbon price for a particular region. However, the large variability of MACC results may indicate a certain level uncertainty. Similar to IPCC climate predictions that attempt to simulate the evolution of future climate, MACCs are approximations and hence probabilistic in their nature. MACC studies should acknowledge this fact and accordingly present MAC of single mitigation options as probabilities (Figure 7.3 and Figure 7.4); but this requires a form of uncertainty assessment that has not yet been undertaken by any MACC study. Communication of uncertainty is important since MACC are tools to directly support mitigation policy design. Failing to communicate uncertainties of MAC estimates poses the danger of informing policy makers incorrectly. This could lead to wrong policy decisions as large-scale implementation of possibly cost-inefficient mitigation options could be enforced by legislation, while more cost-efficient are not implemented. This has particularly severe effects for the innovator country of mitigation policies in the agricultural sector as it cannot draw conclusions from others (Elofsson, 2007). Further, the policy maker runs the risk of failing in executing its responsibility for the public and it's party if the implemented mitigation policy does not result in predicted outcomes. Presenting MACs as probabilities could make the policy maker aware of the expected risks and therefore substantially support the policy design. However, uncertainty is a fundamental part of every policy decision making process and does not only include scientific uncertainty (Knaggård, 2014).

As it could be shown that various ENG and hybrid MACCs are reported for the European agricultural sector, but these predictions are currently under-utilised in the European climate change mitigation agenda. There is a lack of transmission of knowledge production to knowledge utilisation from policy makers and simply increasing knowledge production will not increase its use (Lemos et al., 2012). Decision making support is a social process that depends on several factors, institutions and resources. These are: i) technical factors e.g. information availability, ii) cognitive factors e.g. precipitation of scientific information including accessibility, communication and reliability, iii) institutions that facilitate or limit use of scientific knowledge and iv) structural factors that determine the willingness of the decision makers to use the knowledge (Lemos and Rood, 2010). As with climate predictions, the systematic communication of uncertainty, followed by its reduction can increase the usability of MACC predictions as it addresses possible technical and cognitive barriers i.e. knowledge availability and reliability (Pidgeon and Fischhoff, 2011; Lemos et al., 2012; Weaver et al., 2013); particularly in the agricultural sector as law strongly regulates it. However, policy decisions do not always follow the most rational solution due to the complex nature of policy design (Knaggård, 2014); consequently there are also other factors for under-utilisation of MACC results that may not be within the reach of scientists.

It is the social responsibility of scientists to report and attempt to reduce uncertainty (but some uncertainties are irreducible) in a way that increases scientific influence on political decisions and hence benefits society. However, it is important to distinguish between two locations of uncertainty i.e. in the production process of knowledge and in the communication to the users (Einserink et al., 2013). Latter requires framing complex knowledge about uncertainty in a simple way to be available for policy makers since political agencies lack resources and expertise to adequately manage complex knowledge. Generally, boundary organisations transfer scientific knowledge to decision makers including communication, mediation, translation, feedback and trust building. Sustaining the link between scientists and these boundary organisations could be one way to improve the efficiency of knowledge transfer (Weaver et al., 2013). Similar to the IPCC reports that claim the danger of exceeding a threshold of global temperature increase by more than 2°C, producers of MACCs inevitably take the role of knowledge brokers. Hence, they should directly interact with knowledge users to meet their requirements and communication of uncertainty can increase this interaction. However, climate change mitigation is a strongly politicised topic and this makes it more difficult to communicate risks. The policy maker as potential knowledge user is driven by short-term concerns, typically not much beyond the next elections and therefore it is difficult to communicate uncertainties beyond this time horizon (Einserink et al., 2013).

For effective guidance of policy design, uncertainty assessment of scientists should be undertaken in two different ways. First, scientists report agreement or disagreement of their results within the scientific literature (Funke and Paetz, 2011), ideally with standardised and systematic surveys to measure scientific opinion (Javeline and Shufeldt, 2014). Second, scientists perform uncertainty assessments within their studies e.g. MC analysis. The study of Raadgever et al. (2011) developed a classification of further uncertainty management strategies applied to a water management project. These include i) ignoring the uncertainty which can be with or without awareness; ii) generation of knowledge to assess the uncertainties and reduce them as far as possible; iii) interaction with others including communicating the knowledge of uncertainties and reducing the ambiguity about the system to be managed and iv) coping with uncertainty, particularly with those that cannot be reduced and develop solutions that can cope with different possible outcomes. Such a categorisation could also be developed for MACC studies. Thereby it can be identified which strategies should addressed by whom and how to increase a synergetic effect of different uncertainty management strategies. This would further lead to a distribution of responsibility of uncertainty reporting and operation accordingly.

However, the question arises as to why uncertainties are rarely assessed in the MACC literature. First, expression of uncertainty is always problematic in the light of simultaneously maintaining credibility. Scientists fear facing major criticism e.g. why the model design was chosen despite prior knowledge about uncertainties (Ascher, 2004). More extremely, scientists may think that uncertainties lead to an erosion of public and political trust in their results as scientific uncertainty can be judged as scientific failure from not well-informed knowledge users. This is particularly important for the personal interests of the scientists as it can lead to severe negative impacts on future research funding, personal reputation and pursuing preferred research activities (Ascher, 2004), whereas scientists' dependency on external funding is the key factor to disguise uncertainty. Second, in the light of policy decision guidance, expressing uncertainty could counteract the original purpose. Politicians and interest groups may downplay uncertain scientific results to gain benefits that are against mainstream scientific opinion (Ascher, 2004). Third, public and policy makers assume that the natural world follows deterministic pattern and that science can simulate these 100% accurately. Thereby, they forget that many events are only stochastic in nature e.g. extreme weather events or economic shocks, and this cannot be predicted with absolute certainty (Ascher, 2004). Fourth, uncertainty assessment is a difficult exercise for the scientists as model specific uncertainties are not well known e.g. in case of MC simulation correct pdfs for input parameters are not always available. Such an assessment therefore requires massive resource inputs in terms of working hours and computer process power that may not be available to scientists. Fifth, additional research is required to understand sources of uncertainty (i.e. model parameter, structure, and output, system variability, decision making criteria and linguistic interpretation), the effect of these uncertainties on environmental decision making and possible integration of these sources (Ascough et al., 2008). Finally, uncertainty of scientific results may force the scientists to improve model simulations drastically but in interdisciplinary and complex scientific exercises such as MACC simulation, this is only achievable in a long term horizon with support from adequate resources.

Despite the high level of uncertainty with regard to future climate change prediction reported by the IPCC reports, the majority of policy makers trusts these scientific predictions and is willing to act accordingly and this should convince MACC scientists to communicate uncertainties. The strongly positive attitude towards scientific results and their contribution to society as reported by the National Science Board (2014) should further strengthen researchers' willingness to report uncertainties of their estimates. Communication of uncertainty will also enable to explore ways to increase certainty and in the light of the urgency of climate change mitigation, this could be an opportunity to attract further funding for improving scientific methodologies. Generally, even high levels of model uncertainties regarding cost-efficient GHG reduction cannot justify any inaction in terms of climate change mitigation as inaction would lead to irreversible negative impacts that are certainly larger than consequences of cost-inefficient mitigation strategies (Lemos and Rood, 2010).

7.5.3 Limitations and further improvements

This study is an initial attempt in assessing uncertainty of MAC estimates across the ENG and hybrid MACC literature for the agricultural sector, and more in-depth research is necessary to fully quantify uncertainty of MAC estimates. MACCs contain many uncertainties and uncertainties are cumulating across the MACC development. As discussed earlier scientific literature is missing that attempts to assess or even quantify uncertainty of MACCs. In this light, any contribution to increase the knowledge of MACC uncertainty is an important contribution to the development of the MACC methodology. A meta-analysis is an interesting and popular approach to understand the range of results across literature. It allows identifying gaps in previous research gives a prospect on how MACCs should be developed in future and shows the necessity of reporting in a transparent manner. Therefore, it was decided to utilise this methodology. There is still a lot of work to be done in this field and this research explores an unique way of identifying uncertainties in MACCs. However, given the small number of studies that reported only a few observations, it is likely that the full range of uncertainty could not be captured. The available studies could also be biased in terms of their representativeness e.g. towards particular geographical regions. More observations are required that could enable doing a meta-regression analysis to statistically test for influences on MAC estimates from different study designs. The wide range of differences between MACC study designs is a drawback in our methodology as data harmonisation was limited, but different study characteristics certainly had an impact on quantifying the whole range of uncertainty.

A gap in reported standard errors from each study is a major issue and is a further drawback for this meta-analysis as well as for the KDE approach, as the study observations could not be weighted by their within study estimates of precisions. For future uncertainty assessments, literature should report pdfs of MAC estimates. A future approach to MACC simulation could involve full probabilistic modelling. In the context of climate modelling, the 2009 UK Climate Projections (UKCP09) was an initial approach to estimate uncertainties based on probabilities of climate change scenarios rather than reporting a limited set of equally possible scenarios (Slingo and Palmer, 2011). However, this large-scale approach for the UK also illustrates reasons of why full quantification of uncertainty is rarely undertaken as it requires substantial resource input e.g. working hours and computational resources over an extended period. However, despite these efforts UKCO09 has been under a scanner for not capturing all sources of uncertainty.

7.6 Conclusion

This chapter leads to the following conclusions. There are no less than 19 studies on ENG and hybrid MACCs reporting for single mitigation options in the agricultural sector; of these only 6 are peer-reviewed. Study characteristics and design differ strongly across the studies and this might be a key factor for the scientific discrepancy shown by an extremely high MAC variability.

The eight most often assessed mitigation options are REDFERT, SPLITFERT, TIMEFERT, NOTILL, NITR, AD-E, AD-H, CAD. The mean and mode of these 8 mitigation options revealed that TIMEFERT and NOTILL are the most cost-effective mitigation options. There are good reasons to visualise pdfs of MAC for individual mitigation options but the low number of observations for some mitigation options is a drawback for pdf generation via

KDE. It was shown that a higher study quality and Europe focused MACCs lead to lower CE for some mitigation options; but the uncertainty of the expected outcome remains high.

The ranking of mitigation options based on the unweighted and weighted datasets revealed that increasing soil C storage (only represented by one mitigation option) and reducing soil N_2O emissions are likely to be more cost-effective as compared to manure management measures. Our results showed that despite a large uncertainty, MACCs can guide or advise policy makers about CE of mitigation options. For the agricultural sector, the measures with highest probability to increase benefits while reducing GHG emissions are NOTILL and REDFERT; NITR and AD-H are probably the least cost-efficient measures (in case of both data sets).

The large diversity of MACC methodologies is a drawback for generating more certainty in MACCs. This could be overcome by an uniform MAC estimation methodology; leading to more harmonised estimations and more definitive outcomes that make implications of measure implementation more comparable between MACC studies. Finally, MACC scientists should prioritise quantification of MAC uncertainty and highlight the fact that MAC estimates are probabilistic in their nature. It was shown that revealing uncertainty is not a drawback for scientists and it may lead to increased research funding and faith in scientific results for the urgent issue of climate change mitigation.

Chapter 8 – Conclusion

Climate change is probably the biggest threat to mankind with irreversible impacts on social, economic and ecological systems globally (Stern, 2007). Policy makers globally seek for climate change mitigation and enforced binding GHG reduction on international and national levels. The UNFCCC proposed to limit global temperature increase to a maximum of 2°C to prevent dramatic consequences of climate change but this requires an immediate and significant cut of anthropogenic GHG emissions. Accordingly the EU is aiming to reduce GHG emissions in 2030 by 40% below 1990 levels and China targets carbon intensity reduction by 40-45% per produced unit of GDP relative to 2005 levels. The livestock sector is the main emitter of total methane (CH_4) and nitrous oxide (N_2O), both of which are key GHG (Gerber et al., 2013a). Despite being a major contributor to climate change, the livestock sector is highly vulnerable to a changing climate. However, production levels and GHG emissions from the livestock sector are expected to increase due to an increasing population and demand for livestock products globally. Therefore, livestock production holds a critical role in meeting the urgent target of climate change mitigation. The analysis in this dissertation shows that the EU-15 dairy and Chinese livestock sectors are significant GHG emission sources, and the necessity to reduce GHG emissions from both, particularly in China as GHG emissions from livestock are estimated to increase significantly by 2020 (OECD-FAO, 2013). However, the livestock sector is rarely considered in the global mitigation agenda and is therefore a missed opportunity. To guide policy makers, research must focus on identifying appropriate mitigation options and the economic abatement potentials delivered by livestock sectors globally, including assessment of CE of GHG abatement of various mitigation options. This research study has attempted to fill this knowledge gap by developing two ENG MACCs i.e. for the Chinese livestock sector and EU-15 dairy sector which are exemplary of globally significant livestock production systems. The analysis shows that these sectors offer cost-efficient GHG reduction potentials. The ENG approach was proven to be advantageous in the agricultural sector over other MACC approaches by being measure-specific, selecting more applicable mitigation options, considering the high heterogeneity of the sector and not being limited by assumptions of standard economic theory e.g. production functions or the economy being equal to an equilibrium (Halkos, 2014; Moran et al., 2011).

The current study has emphasised on mitigation options for enteric fermentation based GHG emissions which is the largest source of GHG emissions in ruminant livestock systems. Animal breeding and dietary probiotics proved to be cost negative in both case studies and secondary plant compounds (tea saponins) only in China. However, most of these mitigation options increased production efficiency and thereby decreased GHG emission per unit of product, which is important in the light of globally increasing demand for livestock products. For China, anaerobic digestion was found to offer an immense abatement potential of 58.66 Mt CO_2e at negative costs mainly due to massive subsidies from the Chinese government. Soil C sequestration also offered large abatement potentials with MGI being below a carbon price of 100 ¥/tCO_2e. For EU-15, reduced tillage is also a cost-efficient mitigation option with moderate abatement potentials. Following a review of available MACC literature for the agricultural sector, it was found that mitigation options focussing on agricultural soils are significantly more cost-efficient as compared to measures for manure management.

Further consideration of the policy feasibility is necessary, even for cost-efficient abatement potentials (Smith, 2012). Since MACCs do not account for transaction costs for policy implementation (Kesicki and Ekins, 2012), it is important to identify political instruments that achieve the estimated abatement potentials cost-efficiently (Halkos, 2014). As observed in Figure 3.3, cost-efficient policies can be identified depending on the CE of GHG abatement.

1) For cost negative measures, political instruments include environmental regulation and standards that can enforce mitigation options for GHG reduction directly or indirectly e.g. by feed supplements or animal welfare, respectively.

2) For low cost measures below the carbon price, economic incentive instruments should be applied that include: i) support measures offering financial incentives to the farmers to apply mitigation options which should be strictly regulated to avoid overproduction that may increase negative externalities, ii) emission taxes enforcing taxes to farmers that exceed a certain GHG emission cap or iii) tradable permits where farmers offer certifiable mitigation credits via voluntary carbon trading schemes for buyers outside the sector or hold tradable GHG emission permits within a compulsory carbon trading scheme. For the latter, carbon credits must be verifiable and permanent, which is not possible for all mitigation options.

However, additional costs for high polluting farms impose a competitive disadvantage as compared to producer without carbon payments. To protect regulated livestock production, products that are not subject to carbon pricing could be taxed accordingly.

3) High cost measures are currently too costly for implementation and therefore require investment in research and development from public and private funding, for developing infrastructure or innovation that make these mitigation options more cost-efficient.

To implement these political instruments, carbon accounting tools are required to measure and monitor GHG emissions at farm level throughout the sector. Additionally, education, training and advice services must ensure efficient information dissemination to producers and consumers. Although consumers are neglected in this debate, they hold a key role as driving force for producers to reduce carbon intensities. Failing to inform the farmers about possible mitigation options could lead to non-adoption or adoption of non optimal mitigation options that diminish the target of cost-efficient GHG abatement in the livestock sector. Since such service structures require further development in China and EU-15, only moderate adoption of measures were assumed in this study. However, with a tight grid of information dissemination structures, higher adoption levels can be expected which in turn increases GHG reduction potentials. There is a problem of accounting GHG reduction efforts in GHG inventories and with mitigation efforts being invisible, incentive for policy makers to design mitigation policies is reduced.

This research study showed that MACCs can be a sophisticated tool to inform policy makers about economic abatement potentials, CE of GHG reduction and subsequent prioritisation of mitigation options. While several limitations of the MACC approach were elaborated, emphasis was given to uncertainties related to input and output of the MACC exercise as MACCs have been criticised for the poor treatment of uncertainties. Without assessing input and output uncertainties MACC prediction can be misleading and lead to undesirable mitigation policies. For a policy decision tool it is crucial to increase awareness of potential errors and inform knowledge user about the involved uncertainties (Lempert and Schlesinger, 2000) The projection of the baseline scenario has a crucial importance to the MACC exercise (Paltsev and Capros, 2013), particularly in rapidly changing economies like China (Zhu et al., 2011). In case of China, consideration of different possible baseline scenarios showed significant impacts on MACC results to the extent that animal breeding transformed from a cost negative to a high cost mitigation option. While this shows the importance of predicting the future as accurately as possible, policy makers should be further informed about the implications of different 'what if' scenarios to prevent undesirable political decisions. With reference to the European case study, this dissertation identified that input variables related to adoption potential showed largest contribution to uncertainty of GHG abatement at negative costs. Despite a lack of research in assessment of farmer's willingness to adopt mitigation options and its crucial importance for the MACC exercise, MACCs rely on subjective judgement in this regard. The MC simulation further assessed qualitative uncertainties and showed large uncertainties of measures' CE except for reduced tillage and dietary nitrate. However, ranking of mitigation options by CE of abatement remained stable and remains a valid prioritisation criterion. The MC simulation highlighted the requirement for identifying appropriate pdfs for input variables of the MACC exercise which is currently strongly under-represented in scientific research. By comparing the measures' CE reported by 19 MACC studies for the agricultural sector, it was shown that reported measures' CE varied strongly for same mitigation options, and this also applies for data related to a more homogeneous European agricultural sector. These findings indicate the overall uncertainty of the MACC output. To increase utilisation of MACCs by policy makers, MACC research must prioritise assessment, quantification, report of uncertainties, compare results within the scientific literature and publish data and assumption of the MACC transparently. However, to reduce uncertainties from methodologies and harmonise MACC results from different studies, MACC developer should follow a standardised methodology. Furthermore, MACCs should present the estimates not as finite values as they are rather probabilistic in nature, and report pdfs of MAC estimates. Although scientists may hesitate to reveal uncertainties of their research as they anticipate disadvantages by doing so, uncertainty assessment can be an advantage for the scientists as it can lead to research funding and faith in scientific results, which is particularly important in tackling the urgent issue of climate change mitigation.

To conclude, current shortcomings in the scientific literature that affected this research study will be enumerated to guide future research and enable to strengthen the MACC approach.

1) MACCs usually account for only direct GHG emission within the farm gate. Chapter 5 proved that GHG emissions outside these boundaries significantly contribute to GHG emission output of EU-15 dairy systems and consequently increase abatement potentials of mitigation options. Since climate change is a

global problem, future research should focus on LCA based MACCs for the livestock sector.

2) Prioritising mitigation options solely by their CE can be misleading as the absolute amount of GHG abatement can be neglected. The currently high cost measures could offer cost-efficient GHG abatement in the long term and focusing on currently cheap measures can create a lock-in situation that may hinder implementation of currently high cost measures. As shown in Figure 3.5, MACCs are able to show the correlation between timing and CE of abatement. However, further research must ensure to correct this and facilitate highest possible GHG abatement from the livestock sector.

3) Future research should focus on ancillary benefits and costs of measure implementation including social damage from non climate change pollution and costs for policy implementation, market failures, knowledge dissemination and barriers. Incorporating this information in the MACC exercise could change measures' CE significantly. However, this is a challenge particularly for large regions like China or EU-15.

4) Adoption of new technologies occurs at a slow pace within the livestock sector and policy makers need to account for this fact. For a better understanding of the implications, research should focus on the adoption of new technologies by farmers. This could increase the understanding of implementation barriers and lead to improvement of information dissemination services.

5) In this research study, no interaction factors for simultaneous operating mitigation options were applied since scientific literature does not show robust evidence and these could be subject to strong uncertainties. It is a difficult task to account for interaction of measures in ENG MACCs focussing on large regions. However, future research should supply estimates for e.g. interaction of feed additives. This could have an impact on the overall adoption potentials of mitigation measures.

6) Further research should monitor and evaluate the effect of mitigation options to compare reality with simulation results. This would allow improving future MACC assessments.

7) This research study focussed on mitigation options for production, but demand side mitigation can also show significant reduction potentials, and therefore future research should lay a stronger emphasis on this.

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Appendix

Appendix 1

Alternative MACC approaches

Model-based MACCs

Various types of different model-based approaches have been developed, but this section discusses only the three most common approaches for the agricultural sector i.e. SSM, PEM and CGE. CGE MACCs can be roughly classified as top-down approaches while PEM and SSM MACCs are more suitable to the description of a bottom-up approach. Equilibrium model-based approaches (CGE or PEM) consider a macroeconomic perspective, while SSM MACCs consider the behaviour of a group of farms or even a single farm that are representative to the system of interest. Common for all of these approaches is the fact that they define a set of mitigation options which is fed into a model simulation. Based on standard economic theory, these approaches imply the assumptions of profit maximising behaviour, baseline market being perfectly efficient including perfect information dissemination and other simplified production functions that apply throughout the simulated system. Contrary to SSM, equilibrium approaches further assume market adjustment to equilibrium after system impacts by mitigation option introduction. Implementing an exogenously determined emission reduction target to the simulated system e.g. a carbon tax forces a system response due to which mitigation options are automatically implemented into the system to meet the carbon cap while not exceeding the cost of that carbon tax (Levihn et al., 2014). Based on several targets and subsequent simulation runs, the abatement cost to the economy is estimated, and it increases according to higher levels of GHG reduction targets. Model-based approaches are not measure-explicit; thus the MACC is continuous and without measure specific information (Figure A.1). While SSM MACCs commonly consider in- and output prices as constant parameters, equilibrium approaches account for changes in demand and supply for agricultural products and thus prices are endogenous. For equilibrium approaches, the introduction of a carbon cap therefore directly affects the supply side and indirectly the in- and output prices in the equilibrium (Vermont and De Cara, 2010). Considering market feedbacks is particularly important for assessing the systems' impact after implementation of mitigation options and can have strong influences on the estimated MACCs (Wächter, 2013); and this is of particular importance for MACCs that cover large regions, as changes in supply will have a greater effect on overall prices. Such market feedbacks can also occur through external influences on the system. The study of Morris et al. (2011) serves as an excellent example as it highlights far reaching implications if interacting economies in different countries are considered. This study utilised the Emission Prediction and Policy Analysis (EPPA) model developed at the Massachusetts Institute of Technology to derive a MACC for one country while the economy was influenced by policies of other countries. They conclude that policies from other countries affect prices and thereby baseline development of the country under investigation and not accounting for such influences can cause prediction errors of MACCs. Therefore, it is an advantage for equilibrium approaches to consider market feedbacks and particularly for CGE MACCs to be able to integrate interactions with other economies. However, it is common that CGE MACCs do not account for international trade barriers in a realistic way. SSM MACCs are better equipped for assessment of a microeconomic situation in finer details with regards to the level of disaggregation and heterogeneity of production factors and processes. Hence, this approach can assess measures' impacts in terms of variation of mitigation potential and costs across individual farming systems more accurately as compared to equilibrium model based MACCs (Vermont and de Cara, 2010). Since equilibrium-model based MACCs explore the macroeconomic situation, these are better equipped to assess the full impact of a mitigation policy (Vermont and De Cara, 2010). Both of these approaches are good in accounting for interaction between mitigation options and measures' impact on different GHG emission sources. In this context, SSM MACCs are better equipped for simulating interactions of mitigation options at the farm level e.g. through changes in animal diets. Equilibrium-model approaches are better equipped for capturing interactions being caused by market response after measure implementation. Both approaches can accurately simulate the impact of mitigation options that lead to changes in resource allocation between sectors' activities and do not significantly change the technology of production processes (Vermont and De Cara, 2010). This includes mitigation options that result in changes in production inputs e.g. reduced fertiliser application, reduced number of animals or changes in animals' feeding regime (Vermont and De Cara, 2010). However, this study also reported that equilibrium models generally show lower MACs for a given reduction target as compared to SSM MACCs; making it difficult to compare the results of different model-based approaches.

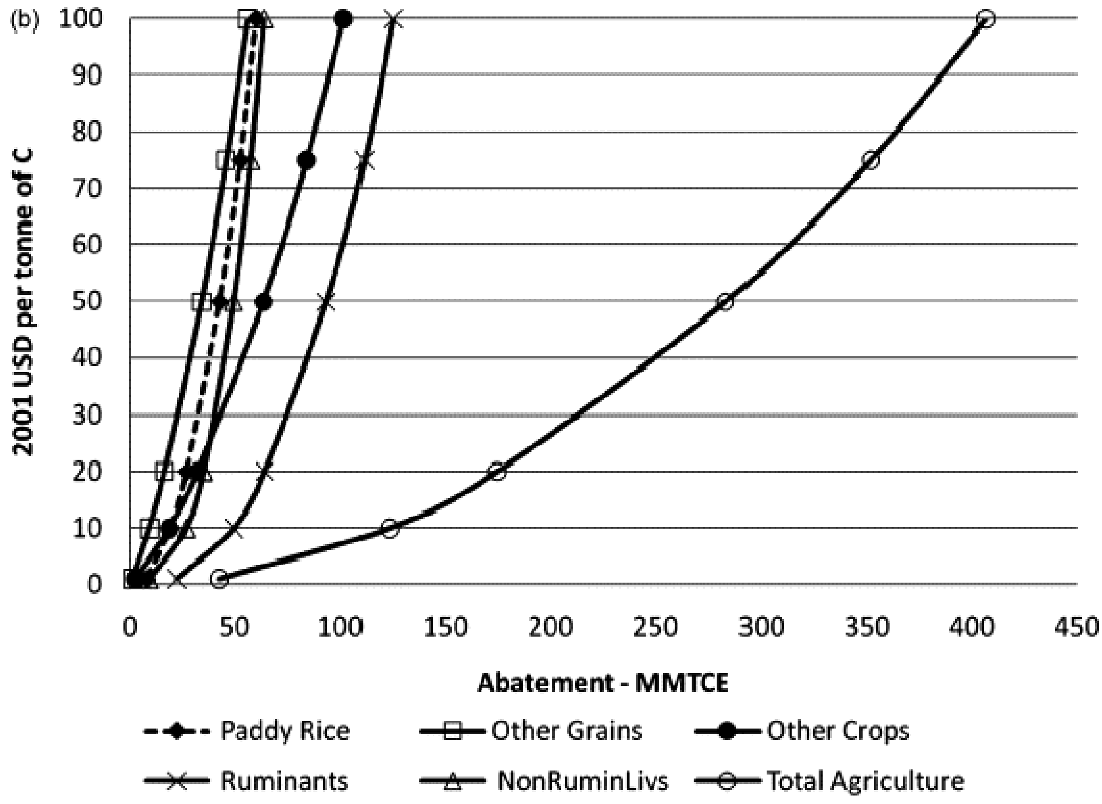

Figure A.1: Example for a CGE MACC on a global level for different agricultural sub-sectors. The y-axis shows GHG abatement in Million Metric Tons of Carbon Equivalent (MMTCE).
Source: Golub et al. (2009)

Advantages and disadvantages of model-based MACCs

The previous section has already discussed and evaluated some characteristics unique to the individual model-based MACC approaches. This section elaborates general pros and cons of the model-based approach.

An advantage of the model-based approach is that inconsistencies in the input data can be easily identified and reduced as compared to ENG MACC approaches. This further facilitates an easy identification of simulation uncertainty. Compared to ENG MACCs, a major advantage is that model-based MACCs are capable of integration of interaction between mitigation options that are implemented simultaneously into the system of interest. As mentioned earlier, this can prevent double counting of GHG abatement. Moreover, consideration of market feedbacks (in case of equilibrium-MACCs) allows automatic incorporation of some behavioural changes in the simulation and this can be a more robust methodology as compared to the manual input in ENG MACCs. A model simulation additionally allows a more consistent baseline and mitigation scenario as compared to ENG MACCs, and this can reduce overall uncertainty.

A disadvantage of model-based MACC approach is the assumption of a market that corresponds to an optimum which may not reflect reality. Major critics against the equilibrium theory were raised and have since been long debated on the added value of standard economic theory. Such assumptions make MACC assessments particularly in developing countries problematic as the economy is to a lesser extent comparable to an optimal market in terms of profit maximising behaviour, perfect information dissemination and competitive market dynamics. Income disparities and structural variation in the agricultural sector are larger in developing countries as compared to developed countries which further erode the argument of a market at equilibrium. In fact, the assumption of an optimal market is questionable in all markets that are distorted by subsidies. This is particularly important for agricultural sectors in many regions e.g. in Europe and China as governmental subsidies are massive. These issues make it difficult or even impossible to create a simulation of non-free markets and markets in developing countries that can trigger rapid structural changes in the economy. An optimal market further rules out the existence of negative cost measures but as shown in the previous section, there are good reasons that negative cost abatement potential exists. The lack of measure specific information in the graphical design is also a disadvantage. Finally, model-based MACCs do not report underlying assumptions and model specification transparently, while ENG MACCs usually report these in a more detailed manner.

Hybrid MACCs

The study of Böhringer and Rutherford (2008) differentiates three types of hybrid MACCs: i) combined MACCs from independently derived bottom-up and top-down simulations, ii) a bottom-up approach that includes macroeconomic feedback or top-down models considering measure specific technological details and iii) totally integrated models based on solution algorithms. One advantage is certainly the ability to report results for individual mitigation options. Further, interaction can be more easily simulated as compared to ENG MACCs. However, due to the model-based approach, this methodology is inherent to most of its shortcomings.

Appendix 2

Alternative tools for uncertainty assessment

Various definitions for uncertainty assessment methodologies are available, and these sometimes show strong differences. Therefore, for the following descriptions of uncertainty assessment tools only the most common characteristics are reported.

Expert elicitation

In cases where model input uncertainties are not known and cannot be derived mathematically, uncertainties must be derived by expert elicitation. This methodology involves a range of different techniques and is a systematic process of gathering, quantifying and structuring judgement from single experts or expert groups on uncertainties preferably in a probabilistic context (Bastin et al., 2013). The procedure usually involves several steps: i) identification and selection of experts, ii) expert consultation including raising concern about subjectivity of judgements, introducing to underlying uncertainties and the elicitation procedure; iii) defining quantification methodologies of uncertainty; iv) evaluating the expert knowledge about uncertainty, particularly advantages and limitations of each expert opinion; v) identification of pdfs, extremes of the pdfs and evaluating the 'accuracy' of the pdfs; vi) verification of the pdfs by experts; and vii) excluding non-useful expert judgements and aggregate the remaining into final pdfs (Refsgaard et al., 2007). The aggregation during the final step can be either based on mathematical methodologies or behavioural aggregation where latter requires interaction of the experts (Bastin et al., 2013). A variation of this methodology is expert judgement that utilises, similar to expert elicitation, judgement of one or several experts for identifying pdfs for future scenarios and impacts of implementation of e.g. new technologies (Willows et al., 2003).

The key advantage of expert elicitation is that it allows the utilisation of a broad range of knowledge that may not be available somewhere else. However, being based on subjective judgement is a major limitation as it involves bias. Further, selection of experts is also a subjective matter and can further increase bias. Therefore, the obtained pdfs are rather suitable for preliminary uncertainty assessment (Refsgaard et al., 2007).

Inverse modelling

Inverse modelling is a single parameter estimation technique where parameter values are optimised by optimising an objective function commonly obtained by deriving the difference between field data and model output (Refsgaard et al., 2007). This results in an optimal or "best fit" calibration of the model. To address the gap of reporting CIs of model output of single point estimation, there are various techniques to estimate post calibration parameter statistics including linear CIs and parameter correlation coefficients (Matott et al., 2009).

A key limitation is that this technique is based on a single model and model structure uncertainty could falsely be allocated to model parameter uncertainty. If the model is used for prediction beyond the calibration base, the identified parameter uncertainty will not compensate for model structure uncertainty (Refsgaard et al., 2007). However, this approach is rather suitable for model-based MACCs.

Multi-Model Analysis

With the help of multiple models focussing on the same system, model structure uncertainty can be assessed. Models can vary for instance in terms of their parameterisation, model codes and boundary conditions. Refsgaard et al. (2006) proposed a framework to utilise the information about model structure error to deal with this uncertainty source. A sub-approach is quantitative Multi-Model Analysis in which each model under evaluation is assigned a performance score for ranking best models and assigning a weight accordingly. The performance score can be based on expert elicitation, stakeholder judgement or quality control procedures (Matott et al., 2009).

An advantage is that the model structure can be evaluated specifically to improve model design and increase robustness of model output. However, the assessor needs to ensure to fully capture the relevant space of plausible models, and that relevant model structures are considered in the analysis (Refsgaard et al., 2007). It can be very time consuming to develop several models of the same system and this limits the efficiency of this approach.

Sensitivity analysis

Sensitivity analysis includes various techniques to evaluate particular model inputs in terms of their impact on model output by changing values for these input variables. This is particularly useful for models with a large number of variables and parameters of which mostly only a few inputs strongly impact the model output (Matott et al., 2009). Sensitivity analysis can be either a fairly simple or complex approach, depending on the model complexity. In case of a linear model output, sensitivity analysis includes a differential analysis that is usually a comparison of the original model output with a model output that is generated by changing one input value and keeping the other constant i.e. a local or 'one factor at a time' sensitivity analysis (Refsgaard et al., 2007). However, possible interactions of model inputs are not considered. Despite being a relatively limited technique, local sensitivity analysis is still widely applied in science (Yang, 2011). In a non-linear model output space, differential analysis applies only for a limited range of model results (Refsgaard et al., 2007). A global sensitivity analysis is in this case more accurate since it covers the total model input and output space and allows for separating particular model output variance that is explained by model inputs and their interaction (Bastin et al., 2013). Global Sensitivity analysis includes analytical methods such as parameter bounding, global screening, variance based, regression-based, correlation based or Monte Carlo based methods (Matott et al., 2009; Yang, 2011).

An advantage of sensitivity analysis is the identification of model inputs that strongly influence the model output. This allows in a second step for ranking and prioritising certain inputs for further uncertainty assessment. However, sensitivity analysis tends to produce a large amount of data and typically does not consider limitations from model structure and system boundaries (Refsgaard et al., 2007).

Appendix 3

Table A.1: Activity levels in the EU-15 dairy sector in 2009 and 2020 by EU member country (considering only specialist dairy farms).

			AT	BL	DK	FI	FR	DE	EL	IR	IT	LU	NL	PT	ES	SE	UK
Dairy farms (1000)		2009	15.3	5.4	4.3	12.8	59.3	60.2	1.3	17.8	32.4	0.6	18.6	7.2	22.8	6.3	15.5
		2020	14.1	4.1	3.0	10.6	52.4	52.5	1.1	14.7	32.4	0.4	16.5	7.2	22.8	6.3	12.2
		Growth (%/yr)	-0.8	-2.7	-3.5	-1.9	-1.2	-1.4	-1.9	-1.9	0	-2.1	-1.2	0	0	0	-2.4
Crops (1000ha)	Permanent grassland	2009	191	126	89.2	10.5	1727	1800	2.3	633	322	22	499	15.7	111	59.8	720
		2020	191	126	89.2	10.5	1727	1806	2.3	633	322	22	499	15.7	111	59.8	720
		Growth (%/yr)	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	Rough grazing	2009	170		2.1	11.8	329	59.0	4.6	109	85.9		21.5	38.2	233	5.9	543
		2020	170		2.1	11.8	329	59.2	4.6	109	85.9		21.5	38.2	233	5.9	543
		Growth (%/yr)	0		0	0	0	0	0	0	0		0	0	0	0	0
	Temporal grassland	2009	17.8	32.4	143	226	904	78.7	2.0	197	188	5.2	145	6.6	15.9	342	255
		2020	19.4	27.0	130	306	809	26.9	1.8	261	180	6.6	212	7.2	21.0	338	198
		Growth (%/yr)	0.9	-1.8	-1.0	3.1	-1.1	-0.2	-1.4	2.9	-0.5	2.4	3.9	0.9	2.8	-0.1	-2.5
	Fodder maize	2009	15.3	48.6	66.1		281	519	0.1	1.0	33.0	0.4	165	10.0	5.0		7.9
		2020	15.3	48.6	66.1		281	517	0.1	1.0	33.0	0.4	165	10.0	5.0		7.9
		Growth (%/yr)	0	0	0		0	-0.0	0	0	0	0	0	0	0		0
	Other fodder	2009	20.2	2.1	59.6	1.4	127	82.3	1.6	6.7	120	2.2	4.5	35.7	56.6	10.6	7.2
		2020	22.1	1.7	54.1	1.8	114	28.2	1.4	8.9	114	2.7	6.6	39.0	74.8	10.4	5.6
		Growth (%/yr)	0.9	-1.8	-1.0	3.1	-1.1	-10.2	-1.4	2.9	-0.5	2.4	3.9	0.9	2.8	-0.1	-2.5
	Feed cereals	2009	20.7	11.6	46.2	140	246	391	0.3	12.4	12.2	7.1	8.3	0.8	11.3	52.8	11.7
		2020	20.9	11.4	46.4	133	245	388	0.3	12.1	11.9	8.9	8.2	0.7	11.4	48.9	13.4
		Growth (%/yr)	0.1	-0.2	0.1	-0.5	-0.1	-0.1	-0.3	-0.2	-0.2	2.2	-0.1	-1.0	0.1	-0.8	1.3
Livestock	Adult cows (M head)	2009	0.35	0.27	0.61	0.44	3.37	3.76	0.09	0.98	1.56	0.04	1.31	0.23	1.01	0.42	2.14
		2020	0.31	0.29	0.53	0.32	2.81	3.10	0.08	1.03	1.41	0.02	1.42	0.18	0.87	0.36	1.82
		Growth (%/yr)	-1.3	0.7	-1.5	-3.2	-1.8	-1.9	-1.7	0.6	-1.0	-4.7	0.86	-2.1	-1.5	-1.7	-1.6
	Young cows (M head)	2009	0.53	0.54	0.86	0.67	6.53	5.07	0.14	2.50	2.24	0.07	1.03	0.36	3.19	0.83	5.40
		2020	0.39	0.43	0.60	0.43	5.58	2.75	0.12	1.92	1.73	0.06	1.02	0.26	2.80	0.48	4.87
		Growth (%/yr)	-3.0	-2.2	-3.5	-4.3	-1.6	-6.0	-1.6	-2.6	-2.6	-2.5	-0.1	-3.1	-1.3	-5.4	-1.0
	Milk yield (1000L/hd)	2009	6.6	6.9	8.3	8.4	6.5	7.4	5.7	5.1	6.5	7.3	7.9	7.1	6.9	8.0	7.1
		2020	7.4	7.7	9.3	9.5	7.3	8.3	6.4	5.7	7.3	8.2	8.8	7.9	7.7	9.0	8.0
		Growth (%/yr)	1.1	1.1	1.1	1.1	1.1	1.1	1.1	1.1	1.1	1.1	1.1	1.1	1.1	1.1	1.1
	Milk production (M t)	2009	2.3	1.9	5.1	3.7	21.9	27.8	0.5	5.0	10.2	0.3	10.3	1.6	7.0	3.4	15.2
		2020	2.3	2.2	4.9	3.0	20.5	25.6	0.5	5.9	10.4	0.2	12.6	1.4	6.7	3.2	14.5
		Growth (%/yr)	-0.1	1.8	-0.4	-2.0	-0.7	-0.8	-0.6	1.7	0.12	-3.7	2.0	-1.0	-0.4	-0.5	-0.5

Table A.2: Input and Output prices in 2009 and 2020. To allow for comparison of the prices, they are mainly shown as costs per ha or per head.

		AT	BL	DK	FI	FR	DE	EL	IR	IT	LU	NL	PT	ES	SE	UK	Source:
								Output price									
Milk price (€/L)	2009	0.30	0.25	0.30	0.39	0.30	0.27	0.39	0.23	0.38	0.27	0.28	0.28	0.30	0.27	0.26	OECD-FAO~
	2020	0.4	0.3	0.4	0.5	0.4	0.4	0.5	0.3	0.5	0.4	0.4	0.4	0.4	0.4	0.4	
	Growth (%/yr)	3	3	3	3	3	3	3	3	3	3	3	3	3	3	3	
Price per cull cow (€/Kg)	2009	2.3	2.6	2.5	2.7	3.1	2.5	2	2.5	2.3	3	2.6	2.2	2	2.2	2.6	EU, commodity price*
	2020	3.5	3.9	4	3.4	4.9	3.9	2.4	4.7	3.9	4.3	4.1	2.8	2.9	4.2	5.1	
	Growth (%/yr)	4.4	4.1	4.7	2.4	4.8	4.8	2.1	6.7	5.1	3.8	4.5	2.5	3.6	6.8	6.9	
								Input price									
Fertiliser (€/ha)	2009	56.0	121.9	90.8	199.2	98.1	95.3	na˙	168.7	89.3	88.2	121.2	141.2	59.4	102.8	140.5	Worldbank†
	2020	72.7	158.5	118.0	258.9	127.5	123.8	na˙	219.3	116.1	114.6	157.6	183.6	77.2	133.6	182.7	
	Growth (%/yr)	2.7	2.7	2.7	2.7	2.7	2.7		2.7	2.7	2.7	2.7	2.7	2.7	2.7	2.7	
Seed and plant (€/ha)	2009	55.2	70.7	95.1	46.2	49.1	62.0	na˙	na	83.6	41.7	50.1	66.2	32.2	42.5	24.2	EU dairy report‡
	2020	65.5	84.0	113	54.9	58.3	73.6	na˙	na	99.3	49.5	59.5	78.6	38.2	50.5	28.8	
	Growth (%/yr)	1.7	1.7	1.7	1.7	1.7	1.7			1.7	1.7	1.7	1.7	1.7	1.7	1.7	
Plant protection (€/ha)	2009	14.9	47.8	43.6	15.3	36.1	42.7	na˙	7.4	51.2	39	29.3	34.5	13.7	12.3	18.4	EU dairy report‡
	2020	17.7	56.8	51.8	18.1	42.9	50.8	na˙	8.8	60.8	46.4	34.8	41.0	16.3	14.6	21.8	
	Growth (%/yr)	1.7	1.7	1.7	1.7	1.7	1.7		1.7	1.7	1.7	1.7	1.7	1.7	1.7	1.7	
Concentrate (€/cow)	2009	287.5	279.8	577.2	650.2	330.2	356.6	na˙	236.5	526.0	243.3	393.7	631.9	641.6	590.6	306.1	Worldbank†
	2020	339.2	330.1	681.1	767.3	389.6	420.8	na˙	279	620.7	287.0	464.6	745.6	757.1	697	361.2	
	Growth (%/yr)	1.7	1.7	1.7	1.7	1.7	1.7		1.7	1.7	1.7	1.7	1.7	1.7	1.7	1.7	
Fodder - purchased (€/cow)	2009	45.3	48.1	143.2	12.7	40.8	22.3	na˙	20.4	107.6	3.5	114.9	56.5	107.7	34.8	215.5	Eurostat‡
	2020	77.0	81.8	243.5	21.5	69.4	37.9	na˙	34.7	182.9	5.9	195.4	96.1	183.1	59.2	366.4	
	Growth (%/yr)	5.4	5.4	5.4	5.4	5.4	5.4		5.4	5.4	5.4	5.4	5.4	5.4	5.4	5.4	
Herd renewal purchases (€/cow)	2009	15.6	23.6	32.5	30.4	15.7	16.5	na˙	38.2	14.5	12.9	13.5	28.3	54.8	40.2	14.7	EU dairy report‡
	2020	18.8	39.5	45.5	44.0	20.3	16.8	na˙	35.1	14.5	31.3	19.5	32.7	101.7	50.6	23.6	
	Growth (%/yr)	1.9	5.3	3.4	3.7	2.6	0.1		-0.8		9.3	3.7	1.4	6.4	2.3	4.8	

~ OECD/FAO (2009-2013): OECD-FAO Agricultural Outlook 2009-2022, ECD Publishing
* Based on historical development stated by European Commission - Agriculture and Rural Development, commodity price monitoring; accessible via http://ec.europa.eu/agriculture/markets-and-prices/price-monitoring/index_en.htm
† The World Bank. (2009-2013). Commodities Price Forecast. World Bank Publications. Online accessible via http://econ.worldbank.org/WBSITE/EXTERNAL/EXTDEC/EXTDECPROSPECTS/0,,contentMDK:21574907~menuPK:7859231~pagePK:64165401~piPK:64165026~theSitePK:476883,00.html
‡ Based on historical development stated by DG-AGri 2010, – EU dairy farms report 2010, European Commission, Brussels

Table A.2: continued

		AT	BL	DK	FI	FR	DE	EL	IR	IT	LU	NL	PT	ES	SE	UK	Source:
							Input price										
Other specific costs (€/cow)	2009	141.4	138.9	254.6	285.3	85.3	170.9	na*	137.1	114.0	160.4	218.0	165.5	128.2	211.0	207.2	EU dairy report‡
	2020	120.5	155.3	322.5	313.6	96.7	187.2	na*	179.3	171.8	197.9	289.7	170.8	165.7	290.1	236.6	
	Growth (%/yr)	-1.6	1.1	2.4	1	1.3	0.9		2.7	4.2	2.1	2.9	0.3	2.6	3.2	1.3	
Machinery and building upkeep (€/cow)	2009	152.9	95.3	228.3	375.2	134.7	153.7	na*	79.3	39.7	175.7	171.0	88.2	77.9	189.9	111.8	EU dairy report‡
	2020	166.1	96.4	284.7	445.8	150.3	158.7	na*	77.5	41.0	193.6	179.0	90.1	110.2	195.5	116.9	
	Growth (%/yr)	0.8	0.1	2.2	1.7	1.1	0.3		-0.2	0.3	1	0.5	0.2	3.5	0.3	0.4	
Energy (fuel and electricity) (€/cow)	2009	136.5	87.8	104.2	249.0	91.1	180.0	62.8	62.1	144.4	125.8	108.7	116.9	82.6	181.0	89.5	Worldbank†
	2020	211.8	136.2	161.7	386.3	141.3	279.2	97.4	96.3	224.1	195.2	168.7	181.3	128.1	280.8	138.8	
	Growth (%/yr)	4.5	4.5	4.5	4.5	4.5	4.5	4.5	4.5	4.5	4.5	4.5	4.5	4.5	4.5	4.5	
Contract work (€/cow)	2009	98.0	131.6	270.3	245.3	194.1	125.8	41.0	70.4	25.1	130.8	161.5	33.7	39.5	154.8	105.4	Expert judgement
	2020	92.8	124.7	256.1	232.4	184.0	119.2	38.9	66.7	23.8	124.0	153.0	31.9	37.4	146.6	99.9	
	Growth (%/yr)	-0.5	-0.5	-0.5	-0.5	-0.5	-0.5	-0.5	-0.5	-0.5	-0.5	-0.5	-0.5	-0.5	-0.5	-0.5	
Other direct inputs (€/cow)	2009	265.7	76.9	152.6	420.1	249.1	188.8	22.3	58.3	87.1	176.8	216.5	49	61	168.9	97.7	EU dairy report‡
	2020	340.7	87.9	100.7	473.7	266.1	181.5	38.1	90.1	80.5	206.4	225.1	45.7	75.8	246.2	88.6	
	Growth (%/yr)	2.5	1.3	-4.1	1.2	0.7	-0.4	5.5	4.4	-0.8	1.6	0.4	-0.7	2.2	3.8	-1	
Wages (€/cow)	2009	12.2	7.3	262.7	130.5	31.3	117.3	50.1	55.7	119.7	38.2	38.1	60.7	32.2	193.1	140.8	Eurostat‡
	2020	13.5	5.9	186.7	120.5	32.0	117.8	45.9	51.2	132.7	34.1	39.2	49.4	36.3	159.6	120.1	
	Growth (%/yr)	0.9	-2.1	-3.4	-0.8	0.2		-0.9	-0.9	1	-1.1	0.3	-2	1.2	-1.9	-1.6	

Appendix 4

Table A.3: Long list of studies focussing on cost of GHG reduction in agriculture.

Study	Included in analysis	Study	Included in analysis	Study	Included in analysis
Amann et al. (2008)	✓	Graus et al. (2004)	✓	Pellerin et al. (2013)	✓
Bates (2001)	✓	Hasegawa and Matsuoka (2010)	✓	Reisinger et al. (2013)	
Bates et al. (2009)	✓	Hediger (2006)		Schneider and McCarl (2006)	
Branca et al. (2012)	✓	Hertel et al. (2009)		Schneider et al. (2007)	
Breen (2008)		Höglund-Isaksson (2012)	✓	Reisinger et al. (2013)	
Briner et al. (2012)		Jacobsen (1998)		Schulte and Donnellan (2012)	✓
Brink et al. (2005)		Kahil and Albiac (2013)	✓	Smith and Upadhyay (2005)	
Creyts et al. 2007		Key and Tallard (2011)		Smith et al. (2008)	✓
de Cara and Jayet (2011)		Koslowski (2015)	✓	USEPA (2006)	✓
de Oliveira Silva et al. (2015)	✓	McCarl and Schneider (2001)		van den Pol-Dasselaar et al. (2013)	✓
de Jager and Blok (1996)		Moran et al. (2008)	✓	van Middelaar et al. (2014)	
Doreau et al. (2014)		Moran et al. (2011)	✓	Vogt-Schilb et al. (2014)	
Durandeau et al. (2010)		Morris et al. (2011)		Wang et al. (2014)	✓
Golub et al. (2009)		Nauclér and Enkvist (2009)	✓	Wagner et al. (2012)	
Gouvello (2010)	✓	O'Brien et al. (2014)		Wetzelaer et al. (2007)	

Example code used in R for the KDE (here for NITR)

```
library (KernSmooth)

library (ks)

dat <- read.csv("XXX.csv")

###########################

tmp <- dat[dat$Measure == "NITR",]

tmp$nc <- NA

for(i in 1:dim(tmp)[1]){

 tmp$nc[i] <- sum(tmp$cluster[i] == tmp$cluster)

}

tmp$wt <- 1/tmp$nc

tmp$wt <- tmp$wt / mean(tmp$wt)

#### unweighted data ####

fit.uncluster <- kde(tmp$CE, eval.points=seq(-1500,1500,1))

getinfo <- function(obj){

  list(mean = sum(obj$eval.points * obj$estimate),

            median = max(obj$eval.points[cumsum(obj$estimate) < 0.5]),

            mode = obj$eval.points[obj$estimate == max(obj$estimate)])

}

getinfo(fit.uncluster)

#### weighted data ####

fit.cluster <- kde(tmp$CE, eval.points=seq(-1500,1500,1), w=tmp$wt)

getinfo <- function(obj){

  list(mean = sum(obj$eval.points * obj$estimate),

            median = max(obj$eval.points[cumsum(obj$estimate) < 0.5]),

            mode = obj$eval.points[obj$estimate == max(obj$estimate)])

}

getinfo(fit.cluster)

#### Study quality ####
```

```
tmpnew <- tmp[tmp$DOC == 1,]

doc.cluster <- kde(tmpnew$CE, eval.points=seq(-1500,1500,1), w=tmpnew$wt)

getinfo <- function(obj){
  list(mean = sum(obj$eval.points * obj$estimate),
             median = max(obj$eval.points[cumsum(obj$estimate) < 0.5]),
             mode = obj$eval.points[obj$estimate == max(obj$estimate)])
}

getinfo(doc.cluster)

#### Location ####

tmpnew <- tmp[tmp$continent == "Europe",]

location.cluster <- kde(tmpnew$CE, eval.points=seq(-1500,1500,1), w=tmpnew$wt)

#### lines(location.cluster$eval.points, location.cluster$estimate, col="green")

getinfo <- function(obj){
  list(mean = sum(obj$eval.points * obj$estimate),
             median = max(obj$eval.points[cumsum(obj$estimate) < 0.5]),
             mode = obj$eval.points[obj$estimate == max(obj$estimate)])
}

getinfo(location.cluster)

#### Cumulative density function ####

plot(fit.uncluster$eval.points, cumsum(fit.uncluster$estimate), type='l', lwd=2, xlab="Cost-effectiveness", ylab="Cumulative probability", xlim=c(-25,250), cex.axis=0.7, cex.lab=0.7)

title(main="NITR", cex.main=0.7)

axis(1,at=seq(-1500,1500,250),labels=seq(-1500,1500), las=2, cex.axis=0.7)

axis(3,at=c(-50000,50000),labels=c("",""), col="white")

axis(4,at=c(-50000,50000),labels=c("",""), col="white")

lines(fit.cluster$eval.points, cumsum(fit.cluster$estimate), lwd=2, col="red")

lines(doc.cluster$eval.points, cumsum(doc.cluster$estimate), lwd=2, col="blue")

lines(location.cluster$eval.points, cumsum(location.cluster$estimate), lwd=2, col="green")

lines(fit.uncluster$eval.points, cumsum(fit.uncluster$estimate), lwd=2, col="black")

lines(c(0,0),c(0,10000),lty=3)
```

```
lines(c(20,20),c(0,10000),lty=3)

lines(c(50,50),c(0,10000),lty=3)

lines(c(100,100),c(0,10000),lty=3)
```

Table A.4: Descriptive statistics for cost-effectiveness of abatement (€/tCO_2e) for each mitigation option

Mitigation option	Mean	Median	Mode	Range	Standard deviation	Minimum	Maximum	No. of observation
REDFERT	52	9	-15	3462	410	-972	2490	109
SPLITFERT	57	3	0	624	158	-72	552	26
TIMEFERT	-4	41		578	114	-488	90	32
NOTILL	-64	-66		467	84	-328	140	26
NITR	97	83	9	418	89	-19	400	57
AD-E	43	19	34	1244	147	-462	782	79
AD-H	168	53	19	1086	255	11	1097	51
CAD	572	348		2429	748	-78	2352	79